PB85-102333

Effluent and Ambient Toxicity Testing and Instream Community Response on the Ottawa River, Lima, Ohio

(U.S.) Environmental Research Lab.-Duluth, MN

Aug 84

U.S. Department of Commerce
National Technical Information Service
NTIS

United States
Environmental Protection
Agency

Permits Division
(EN-336)
Washington DC 20460

Environmental Research
Laboratory
Duluth MN 55804

OWEP 84-01

EPA-600/3-84-080
August 1984

PB85-102333

Research and Development

EPA

Effluent and Ambient Toxicity Testing and Instream Community Response on the Ottawa River, Lima, Ohio

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TECHNICAL REPORT DATA

(Please read Instructions on the reverse before completing)

1. REPORT NO. EPA-600/3-84-080	2.	3. RECIPIENT'S ACCESSION NO. PB85 102333
4. TITLE AND SUBTITLE Effluent and Ambient Toxicity Testing and Instream Community Response on the Ottawa River, Lima, Ohio		5. REPORT DATE August 1984
		6. PERFORMING ORGANIZATION CODE
7. AUTHOR(S) Donald I. Mount, Nelson A. Thomas, Teresa J. Norberg, Michael T. Barbour, Thomas H. Roush, and William F. Brandes		8. PERFORMING ORGANIZATION REPORT NO.
9. PERFORMING ORGANIZATION NAME AND ADDRESS Environmental Research Laboratory 6201 Congdon Boulevard Duluth, MN 55804		10. PROGRAM ELEMENT NO.
		11. CONTRACT/GRANT NO.
12. SPONSORING AGENCY NAME AND ADDRESS Environmental Research Laboratory Office of Research and Development U.S. Environmental Protection Agency Duluth. MN 55804		13. TYPE OF REPORT AND PERIOD COVERED
		14. SPONSORING AGENCY CODE EPA/600/03
15. SUPPLEMENTARY NOTES		

16. ABSTRACT

The research described in this report had three objectives: 1) determination of effluent and ambient stream toxicity to Ceriodaphnia, fathead minnows, and indigenous species; 2) definition of the response of the biological community to point-source discharges; and 3) evaluation of the effectiveness of toxicity testing techniques in predicting ambient toxicity to indigenous communities.

Based on toxicity measurement of effluents and river water, tests did predict the resulting toxicity downstream from the discharges. A correlation was established between ambient toxicity and biological impact which suggests that effluent and ambient toxicity tests are accurate predictors of receiving water impact.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group

18. DISTRIBUTION STATEMENT	19. SECURITY CLASS *(This Report)*	21. NO. OF PAGES 90
	20. SECURITY CLASS *(This page)*	22. PRICE

EPA Form 2220-1 (Rev. 4-77) PREVIOUS EDITION IS OBSOLETE

EPA-600/3-84-080
August 1984

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by

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Contents

Page

List of Contributors iii

List of Tables vii

List of Figures x

Foreword xii

Executive Summary xiv

Quality Assurance xv

1. Introduction 1-1

2. Study Design 2-1

 2.1 Toxicity Testing Study Design 2-1
 2.2 Field Survey Study Design 2-2
 2.3 Approach to Integration of Laboratory and Field Efforts 2-3

3. Site Description 3-1

4. Toxicity of Effluents and Receiving Water, 1982 4-1

 4.1 Chemical/Physical Conditions 4-1
 4.2 Results of Fathead Minnow Growth Test 4-1
 4.3 Results of Reproductive Potential Tests Using *Ceriodaphnia* 4-2
 4.4 Evaluation of Toxicity Impact 4-6

5. Toxicity of Effluents and Receiving Water, 1983 5-1

 5.1 Results 5-1
 5.2 Discussion 5-3

6. Dilution Analysis of the Sewage Treatment Plant, Refinery, and Chemical Plant, 1982 6-1

 6.1 Sewage Treatment Plant 6-1
 6.2 Refinery 6-3
 6.3 Chemical Plant 6-4
 6.4 Evaluation of Dilution Characteristics 6-5

7. Periphytic Community, 1982 Survey 7-1

 7.1 Community Structure 7-1
 7.2 Chlorophyll *a* and Biomass 7-2
 7.3 Evaluation of Periphytic Community Response 7-3

8. Benthic Macroinvertebrate Community, 1982 Survey 8-1

 8.1 Community Structure 8-1
 8.2 Spatial Trends in Key Taxa 8-4
 8.3 Benthic and Zooplankton Drift Collections 8-6
 8.4 Evaluation of the Macroinvertebrate Community 8-7

Contents (Continued)

Page

9. Fish Community, 1982 Survey 9-1
    - 9.1 Community Structure 9-1
    - 9.2 Evaluation of Fish Community Response 9-2
10. Fish Caging Study, 1982 Survey 10-1
    - 10.1 *In Situ* Toxicity Testing 10-1
11. Benthic Macroinvertebrate Community, 1983 Survey 11-1
    - 11.1 Community Structure 11-1
    - 11.2 Spatial Trends of Major Groups 11-1
    - 11.3 Comparison Between 1982 and 1983 Surveys 11-4
12. Fish Community, 1983 Survey 12-1
    - 12.1 Community Structure 12-1
    - 12.2 Comparison Between 1982 and 1983 Surveys 12-1
13. Zooplankton Community, 1983 Survey 13-1
    - 13.1 Community Structure 13-1
    - 13.2 Evaluation of Zooplankton Community Response 13-1
14. Comparison of Laboratory Toxicity Data and Receiving Water Biological Impact 14-1
    - 14.1 Results of Integration Analyses 14-3
    - 14.2 1982-1983 Comparison 14-4
    - 14.3 Calculation of Toxicity Reduction 14-6

References R-1

Appendix A: Toxicity Test Methods A-1

Appendix B: Hydrological Methods B-1

Appendix C: Biological Methods C-1
- C.1 Periphyton C-1
- C.2 Benthos C-1
- C-3 Fisheries C-1
- C.4 Fish Caging Study C-2
- C.5 Zooplankton C-2

C-2

Appendix D: Support Biological Data D-1

List of Tables

Table Page

4-1 Seven-Day Percent Survival of Larval Fathead Minnows Exposed to Various Concentrations of Three Effluents in Upstream Water, Lima, Ohio, 1982 4-2
4-2 Average Concentration of Stream Water and Effluent Below Each Discharge During the 1982 Testing Period 4-2
4-3 Mean Dry Weight of Larval Fathead Minnows Exposed to Three Effluents at Various Concentrations, Lima, Ohio, 1982 4-3
4-4 Seven-Day Percent Survival of Larval Fathead Minnows Exposed to Refinery Waste Diluted with Two Different Dilution Waters, Lima, Ohio, 1982 4-3
4-5 Mean Dry Weight of Larval Fathead Minnows Exposed to Refinery Effluents Diluted with Two Different Dilution Waters, Lima, Ohio, 1982 4-3
4-6 Mean Young Per Original Female and Mean Percent Survival of Adult *Ceriodaphnia* in Various Effluent Concentrations Using Receiving Water for Dilution, Lima, Ohio, 1982 4-4
4-7 Mean Young Per Original Female and Percent Survival of Adult *Ceriodaphnia* in Refinery Effluent Concentrations Using Lake Superior Water for Dilution, Lima, Ohio, 1982 4-4
4-8 Mean Young Production and Percent of Survival of *Ceriodaphnia* for the Ambient Toxicity Tests in 1982 4-5

5-1 Chemistry Data for Three Effluents in Station 1 Water for Fathead Minnow Larval Growth Tests, Lima, Ohio, 1983 5-1
5-2 Water Chemistry Data for Ambient Toxicity Test with Fathead Minnows at Various River Stations, Lima, Ohio, 1983 5-1
5-3 Final Dissolved Oxygen Concentrations for *Ceriodaphnia* Tests on Effluents and Stream Station Water, Lima, Ohio, 1983 5-2
5-4 Seven-Day Percent Survival of Larval Fathead Minnows Exposed to Various Concentrations of Three Effluents in Station 1 Water, Lima, Ohio, 1983 5-2
5-5 Mean Weight of Larval Fathead Minnows Exposed to Three Effluents at Various Concentrations, Lima, Ohio, 1983 5-3
5-6 Seven-Day Percent Survival of Larval Fathead Minnows Exposed to Water from Various Stream Stations for Ambient Toxicity, Lima, Ohio, 1983 5-3
5-7 Mean Weight of Larval Fathead Minnows Exposed to Water from Various Stream Stations for Ambient Toxicity, Lima, Ohio, 1983 5-3
5-8 Mean Young Per Female *Ceriodaphnia* and Mean 7-Day Percent Survival of Original Test Animals Exposed to Various Effluent Dilutions in Station 1 Water, Lima, Ohio, 1983 5-4
5-9 Mean Young Per Female *Ceriodaphnia* and 7-Day Percent Survival of Original Test Animals Exposed to Water from Various Stream Stations for Ambient Toxicity, Lima, Ohio, 1983 5-4

List of Tables (Continued)

Table | | Page
5-10 Geometric Mean of the Effect and No-Effect Concentration for the Three Effluents and Two Test Species, Lima, Ohio, 1983 5-4
5-11 Predicted Concentrations of STP and Refinery Effluent at Near-Field Stations Based on Conductivity Measurements, Lima, Ohio, 1983 5-5
6-1 Transect Locations for the Dye Dilution Analysis at Three Sites on the Ottawa River, 1982 Survey 6-1
6-2 River Flows Upstream of the STP and Reported Discharge Flows at Each Site, 1982 Survey 6-3
6-3 Ottawa River Flow and Percent Flow Contribution from the Discharges on the Days of the Three Dye Surveys, 1982 Survey 6-4
7-1 Summary of Periphyton Composition, Diversity, and Standing Crop on Natural Substrates in the Ottawa River, September 1982 7-1
8-1 Average Density of the Most Abundant Species at Each Sampling Station, Ottawa River, 21 September 1982 8-2
8-2 Density of Macroinvertebrates Collected from the Drift, Ottawa River, 23 September 1982 8-6
9-1 Results of Fisheries Survey of Ottawa River, Abundance by Station, 24-26 September 1982 9-1
10-1 Results of Fish Caging Study, Ottawa River, 1982 Survey 10-1
11-1 Composition of the Benthic Community of the Ottawa River, July 1983 11-2
11-2 Abundance of Benthic Macroinvertebrates Collected from the Ottawa River, July 1983 11-3
12-1 Results of Fish Collections in the Ottawa River, July 1983 12-1
13-1 Planktonic Organisms Collected from the Ottawa River, July 1983 13-1
14-1 Comparison of Toxicity and Biological Response 14-1
C-1 Station Pool, Riffle Proportions, and Number of Seine Hauls C-2
D-1 Abundance of Periphytic Algae on Natural Substrates in the Ottawa River, September 1982 .... D-1
D-2 Chlorophyll *a* and Biomass Data and Statistical Results for Periphyton Collected from Natural Substrates in the Ottawa River, September 1982 .... D-2
D-3 Ranked Abundance Listing of All Macroinvertebrates Collected from Ottawa River, 21 September 1982 .... D-3
D-4 Shannon-Wiener Diversity Indices and Associated Evenness and Redundancy Values and Community Loss Indices Calculated on Benthic Data from Ottawa River, 1982 .... D-5

List of Tables (Continued)

Table **Page**

D-5 List of Fish Species and Families Collected from the Ottawa River Near Lima, Ohio, 24-26 September 1982 . . . . D-5

D-6 Shannon-Wiener Diversity Indices, Associated Evenness and Redundancy Values, and Community Loss Indices Calculated on Fisheries Data from Ottawa River, 1982 . . . . D-6

List of Figures

Figure | Page
2-1 Study area, Ottawa River, Lima, Ohio 2-2
2-2 Histogram of LC50s for copper of fresh water species 2-3
4-1 Fathead minnow growth tests for STP, refinery and chemical effluents, Lima, Ohio, 1982 4-2
4-2 Comparison of fathead minnow weights in refinery effluent using two dilution waters, Lima, Ohio, 1982 4-3
4-3 *Ceriodaphnia* young production in three effluents, Lima, Ohio, 1982 4-4
4-4 *Ceriodaphnia* young production in ambient stream stations, Lima, Ohio, 1982 4-6
5-1 Fathead minnow growth tests for STP, refinery, and chemical effluents, Lima, Ohio, 1983 5-2
5-2 Fathead minnow growth tests for ambient stations, Lima, Ohio, 1983 5-4
5-3 *Ceriodaphnia* young production in three effluents, Lima, Ohio, 1983 5-4
5-4 *Ceriodaphnia* young production for ambient stations, Lima, Ohio, 1983 5-5
6-1 Surface dilution contours in the Ottawa River downstream from the STP, 21 September 1982 6-2
6-2 Bottom dilution contours in the Ottawa River downstream from the STP, 21 September 1982 6-2
6-3 Dilution contours in the Ottawa River downstream from the refinery, 23 September 1982 6-4
6-4 Dilution contours in the Ottawa River downstream from the chemical plant, 25 September 1982 6-5
6-5 Effluent contribution to receiving water 6-6
7-1 Spatial distribution of periphyton community indices and associated parameters, 1982 survey 7-2
7-2 Spatial distribution of key periphyton taxa, 1982 survey 7-4
8-1 Spatial patterns of benthic species diversity and components of diversity, Ottawa River, 1982 survey 8-3
8-2 Spatial abundance patterns of key benthic taxa, Ottawa River, 1982 survey 8-4
8-3 Spatial abundance patterns of the dominant ephemeropterans, Ottawa River, 1982 survey 8-5
8-4 Spatial distribution of major benthic groups, 1982 survey 8-6
8-5 Spatial trends of proportion of population in drift compared to benthic standing crop for major taxonomic groups, 1982 survey 8-7

List of Figures (Continued)

Figure **Page**

9-1 Spatial distribution of fish community indices and associated parameters, 1982 survey 9-3
9-2 Spatial distribution of selected fish species and community parameters, 1982 survey 9-4
10-1 Results of *in situ* fish caging study, Ottawa River, 1982 survey 10-2
11-1 Spatial trends of benthic community parameters, 1983 11-1
11-2 Spatial trend of major benthic taxonomic groups, 1983 11-2
12-1 Spatial trends of selected fish abundances, July 1983 12-2
13-1 Spatial trends of zooplankton components of the plankton, July 1983 13-2
14-1 Correlation of *Ceriodaphnia* young per female with benthic parameters from eight stations in the Ottawa River, Lima, Ohio, 1982 14-3
14-2 Correlation of *Ceriodaphnia* young per female and algal diversity at eight stations in the Ottawa River, Lima, Ohio, 1982 14-3
14-3 Ambient toxicity correlation between *Ceriodaphnia* young per female and ecological survey data for 1982 14-3
14-4 Ambient toxicity correlation between *Ceriodaphnia* young per female and ecological survey data for 1983 14-4
A-1 Test chamber for static renewal fathead minnow larvae growth test A-2

Foreword

This report is the first in a series of reports which present the results of the Complex Effluent Toxicity Testing Program. The program is a multi-year research project conducted by EPA's Environmental Research Laboratory, Duluth, Minnesota, and the Permits Division, Office of Water Enforcement and Permits, Washington, DC. Contractor support was provided by Ecological Analysts, Inc. of Baltimore, Maryland.

The Complex Effluent Toxicity Testing Program was initiated to support the developing trend toward water quality-based toxics control in the National Pollutant Discharge Elimination System (NPDES) permit program. It is designed to investigate, under actual discharge situations, the appropriateness and utility of "whole effluent" toxicity testing in the identification, analysis, and control of adverse water quality impact caused by the discharge of toxic effluents.

Until recently, the focus of NPDES permitting was on achieving technology-based control levels for toxic and conventional pollutants. Regulatory authorities set permit limits on the basis of national guidelines. Control levels reflected the best treatment technology available considering technical and economic achievability. Such limits did not, nor were they designed to, protect water quality on a site-specific basis.

The NPDES permits program, in existence for over 10 years, has achieved the goal of the implementation of technology-based controls. With these controls largely in place, future controls for toxic pollutants will, of necessity, be based on site-specific water quality considerations. Unfortunately, there has been less practical experience in setting water quality-based controls, particularly for toxicants, than in setting technology-based controls.

Setting water quality-based controls for toxicants can be accomplished in two ways. The first is the pollutant-specific approach which involves setting limits for single chemicals based on laboratory derived no-effect levels. The second is the "whole effluent" approach which involves setting limits using effluent toxicity as a control parameter. There are advantages and disadvantages to both approaches.

The major advantages of the pollutant-specific approach include the capability to design treatment for specific chemicals in an effluent and the availability of large amounts of data on the effects of some individual toxicants. Although these data are available for only a small number of toxicants, the regulatory authorities are familiar with it and use it extensively. The disadvantages of the approach are that it cannot cover the large number of potentially toxic pollutants in wastestreams, the effects of mixtures cannot be determined, and the bioavailability of individual chemicals at different discharge sites cannot be ascertained. For the whole effluent approach, the major advantages include the capacity to analyze the combined toxicity of all constituents of complex effluents and the ability to measure the bioavailability of those constituents. The major disadvantages are that there is a lack of toxicity treatability data and permitting experience. Regulatory authorities are unfamiliar with this approach.

Effluent toxicity has not been used as an effluent control parameter in NPDES permitting despite its practicality potential. Even many water quality specialists have been hesitant to employ toxicity as a control parameter. There are two reasons which can be identified. First, toxicity tests have been considered too imprecise and too variable to be used in controlling toxics in permits. Second, there is no effluent toxicity database available to indicate what measured effluent toxicity really means in terms of receiving water impact. There has been considerable criticism directed at toxicity testing as a permitting tool because it was thought that the results of effluent toxicity testing did not translate to instream biological impact.

The Complex Effluent Toxicity Testing Program has been designed to analyze these concerns. The program has four major purposes:

1. To investigate the validity of effluent toxicity tests to predict and quantify adverse impact to receiving waters caused by the discharge of toxic industrial and municipal effluents;
2. To determine appropriate testing procedures which will support regulatory agencies as they begin to establish water quality-based toxics control programs;
3. To serve as practical case examples of how such testing procedures can be applied in different toxic effluent discharge situations involving single and multiple discharges in a variety of dilution situations; and
4. To field test recently developed short-term chronic toxicity tests involving the test organisms *Ceriodaphnia* and *Pimephales promelas*.

Study sites were selected on the basis of several selection criteria. The primary criterion was that the receiving water had to have a history of adverse water quality impact which was associated with existing NPDES discharges. Other selection criteria included relatively low available dilution at low flow periods, the presence of a mix of industrial and municipal dischargers, the probability of the discharges containing toxic pollutants at toxic levels, and the cooperation of both the regulatory agencies and the regulated community.

To date, six sites have been investigated involving 11 municipal discharges and about 30 industrial discharges. They are, in order of investigation:

1. Ottawa River, Lima, Ohio
2. Scippo Creek (tributary to the Scioto River), Circleville, Ohio
3. Five Mile Creek, Birmingham, Alabama
4. Skeleton Creek, Enid, Oklahoma
5. Naugatuck River, Waterbury, Connecticut
6. Patapsco and Back Rivers, Baltimore, Maryland

The Lima, Ohio site and the Birmingham, Alabama site were subjected to further analysis for comparative purposes one year after the initial site visits were conducted. Two more sites involving larger rivers and estuaries are planned for study during 1984.

This project is a research effort only and has not involved either NPDES permit issuance or enforcement activities. Toxicity testing of the effluents was conducted for research purposes only.

The following report describes the Lima, Ohio study conducted in September 1982 and July 1983.

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Project Officers
Complex Effluent Toxicity
Testing Program

Executive Summary

EPA recently issued a policy which provides for control of the discharge of toxic substances through the use of numerical criteria and effluent toxicity limits in NPDES permits. This is the first broad scale effort to use effluent toxicity in the NPDES permit program and a scientific basis for this approach is needed.

The research on the Ottawa River described in this report had three objectives:

1. Determination of effluent and ambient stream toxicity to *Ceriodaphnia*, fathead minnows, and indigenous species.
2. Definition of the response of the biological community to point-source discharges.
3. Evaluation of the effectiveness of toxicity testing techniques in predicting ambient toxicity to indigenous communities.

The Ottawa River at Lima, Ohio receives discharges from the municipal sewage treatment plant (STP), a refinery, and a chemical company. In addition to the three effluents, toxicity tests were conducted on samples from 13 river stations. Biological studies were conducted at eight of the stations and included benthos, fish, algae, and zooplankton. Studies were conducted in 1982 and 1983 to assess the reproducibility of methodologies and results.

The STP effluent was toxic to *Ceriodaphnia* but not to fish. The river downstream from the STP was toxic and there was a severe biological impact in that same area. The refinery effluent was toxic to both test organisms. The river was also toxic downstream from both the refinery and the chemical plant. Both the benthic and fish communities were severely altered in these areas. The biological impact ended at the same stations as those having no toxicity in the lab tests.

Based on toxicity measurement of effluents and river water, toxicity tests did predict the resulting toxicity downstream from the discharges. A correlation was established between ambient toxicity, effluent toxicity and biological impact which suggests that effluent and ambient toxicity tests are accurate predictors of receiving water impact.

Quality Assurance

Coordination of the various studies was completed by the principal investigator preceding and during the onsite work. A reconnaissance trip was made to the site before the study and necessary details regarding transfer of samples, specific sampling sites, dates of collections, and measurements to be made on each sample were delineated. The evening before the study began, a meeting was held onsite to clarify again specific responsibilities and make last minute adjustments in schedules and measurements. The mobile laboratory was established as the center for resolution of problems and adjustment of work schedules as delays or weather affected the completion of the study plans. The principal investigator was responsible for all Quality Assurance related decisions onsite.

All instruments were calibrated by the methods provided by the manufacturers. For sampling and toxicity testing, the protocols described in the referenced published reports were followed. Where identical measurements were made in the field and laboratory, both instruments were cross-calibrated for consistency.

1. *Introduction*

To date, the focus of water pollution control in the National Pollutant Discharge Elimination System (NPDES) permits program has been on the attainment of national technology requirements and the implementation of water quality criteria for the 129 "priority pollutants." However, implementation of these standards and criteria does not always guarantee that certain dischargers will not cause adverse effects to receiving waters. Industrial and municipal effluents often contain large numbers of potentially toxic pollutants which can move through treatment systems virtually untreated. Often these are pollutants for which little or no toxicity data exist. Further complications arise from the potential interaction of combinations of pollutants to increase or decrease toxicity.

Future activities in water pollution control will focus on the control of toxic pollutants which impact water quality. There are two methods used in controlling toxic impact: pollutant-specific controls and "whole effluent" toxicity-based controls. Because toxicity testing evaluates a living organism's response, it has an advantage over chemical-specific analyses which may not identify all pollutants in a wastewater sample and which cannot detect toxicity interactions. Toxicity information can indicate the need for additional characterization of an effluent and can also provide a basis for permit limits based on state water quality standards for toxicity- or technology-based requirements.

The primary purpose of this study is to investigate the relationship between effluent toxicity data and ecological response. Thus, three objectives must be met:

1. Determination of effluent and ambient stream toxicity to *Ceriodaphnia*,[a] fathead minnows, and indigenous species.
2. Definition of the response of the ecological community to point-source dischargers.
3. Evaluation of the effectiveness of toxicity testing techniques in predicting ambient toxicity to indigenous communities.

This report is organized into sections corresponding to the project tasks. Following an overview of the study design and a summary of the description of the site, the chapters are arranged into toxicity testing, hydrology, and ecological surveys for the two study periods (September 1982 and July 1983). An integration of the laboratory and field studies is presented in Chapter 14. All methods and support data are included in the appendixes for reference.

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[a] The species used in the 1982 study was *Ceriodaphnia reticulata*. The species of *Ceriodaphnia* used for these tests is not known with certainty. The stocks were thought to be *C. reticulata* but, in November 1983, based on taxonomic verification by Dorothy Berner, Ph.D. (Temple University, PA), a second species, *C. affinis/dubia*, was also discovered in the stock cultures. The exact determination of the species tested is not critical to this study, and all reference is to the genus only in this report.

2. Study Design

The primary purpose of this study was to investigate the ability of laboratory effluent toxicity tests to predict ambient stream toxicity impacts at a multiple discharge site on a small river system. The site chosen for study was the Ottawa River near Lima, Ohio. The study area included three dischargers: a sewage treatment plant (STP), a refinery, and a chemical plant. A more complete description of the study area is found in Chapter 3. This study required laboratory tests that centered on expected effluent dilution concentrations and organisms that had toxicity sensitivitiy similar to indigenous stream organisms. In conjunction with these toxicity tests, ecological surveys of the Ottawa River were conducted to identify structural effects to representative biotic communities and selected populations from point source discharges. Hydrological analyses included effluent configuration studies to define the mixing and interactive characteristics of the effluents. The results from all of these study components were then compared. Thus, the study of the Ottawa River (Figure 2-1) at Lima, Ohio, consisted of four parts:

1. Effluent and ambient toxicity testing using the 7-day *Ceriodaphnia,* fathead minnow, and indigenous species test,
2. Biological community characterization,
3. Hydrological measurements, and
4. Integration and interpretation of results.

The study was conducted initially during 21-28 September 1982. A follow-up study was conducted 7-8 July 1983 following an operational modification by the refinery. The methods used in the study during the two time periods are detailed in Appendixes A, B, and C. The respective study designs for the laboratory and field aspects as well as the data analysis task are outlined in the following sections.

2.1 Toxicity Testing Study Design

Toxicity tests were performed on each of the three effluents to measure subchronic effects on growth of larval fathead minnows and chronic reproductive effects on *Ceriodaphnia*. A range of effluent concentrations was used so that acute mortality also could be measured, if it existed. Acute toxicity is defined as short-term effects with lethality as the endpoint. Chronic toxicity is considered long term (length of time is dependent upon test species), where both lethal and sublethal effects are considered. For these tests, acute mortality is usually measured at 48 hours for *Ceriodaphnia* and 96 hours for fathead minnows. The objective of these tests therefore was to estimate the minimum concentration of each effluent that would cause acute mortality and chronic effects on growth (fathead minnows) or reproduction (*Ceriodaphnia*).

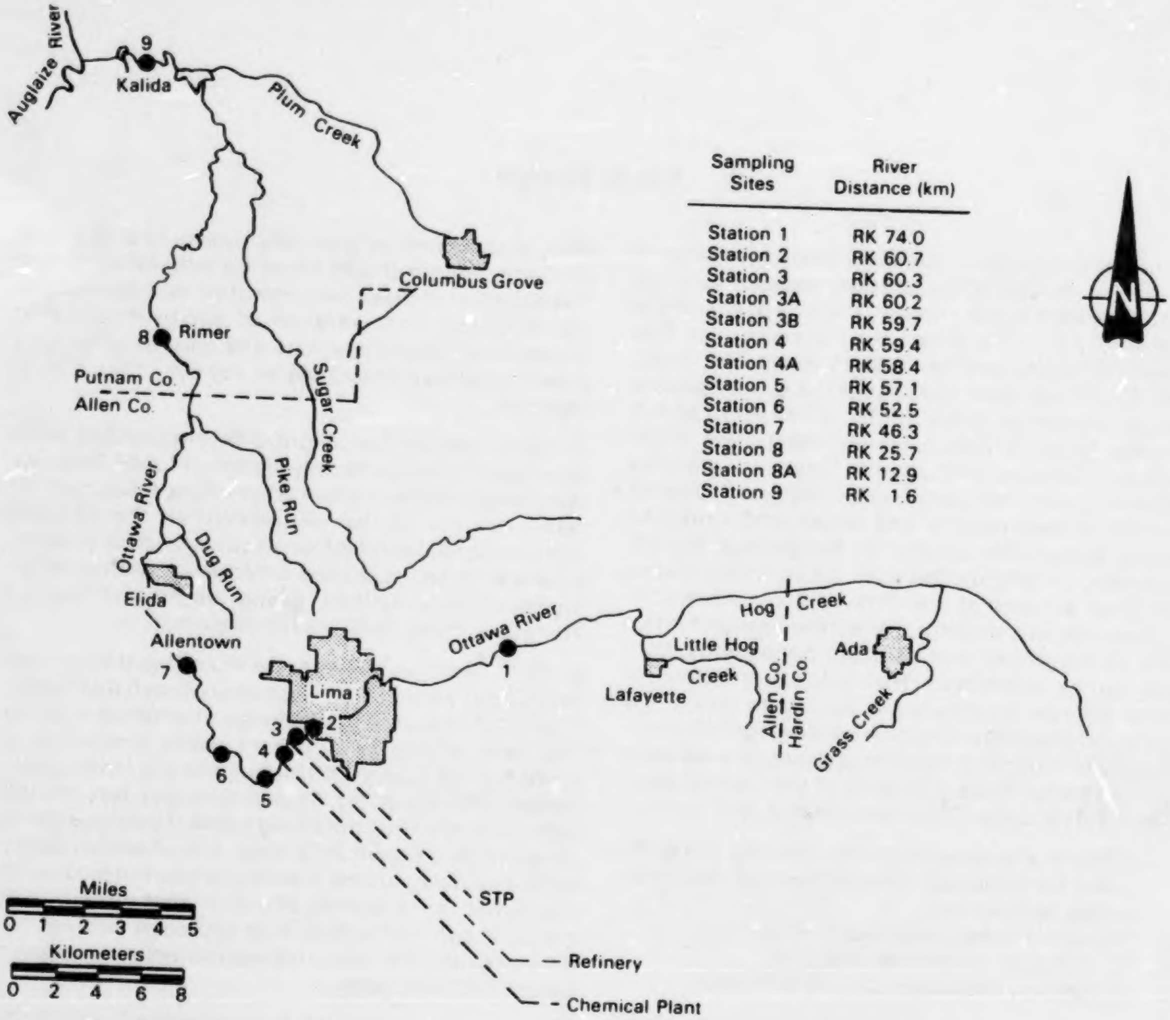
Resident species from eight different families were also tested for acute toxicity of each effluent. This was done to determine if there were any species more or less resistant to the effluents than the fathead minnows and *Ceriodaphnia* used in the chronic tests. However, many problems inherent in testing indigenous species resulted, giving invalid test results. Therefore, these data are not presented.

In 1982, the dilution water for the effluent tests was taken from immediately upstream of each discharge. Therefore, the second discharge downstream of the first was diluted with stream water containing a portion of the upstream effluent, and the most downstream effluent of the three discharges was diluted with stream water containing some of both upstream effluents (see Figure 2-1). Thus, the inherent toxicity of the two downstream discharges was not measured but rather the combined effects of that effluent and the upstream effluent(s). This approach was necessary because the objective was to estimate impact below each discharge.

A separate set of tests was performed with both *Ceriodaphnia* and fathead minnows on the refinery effluent in which a high quality dilution water (Lake Superior water) was used in order to measure only the inherent toxicity of the refinery waste. Another *Ceriodaphnia* test was conducted using unchlorinated STP effluent and diluent water from above the City of Lima. The purpose of this test was to determine if any toxicity observed was due to chlorination or contributions from runoff from the City of Lima.

In addition to the above tests (hereafter referred to as the effluent dilution [ED] tests), stream stations were established from above the discharges extending downstream to just before the confluence with the Auglaize River to measure ambient toxicity. The purpose of these tests was to measure the loss of toxicity of the effluents after mixing, dilution from other stream inputs, degradation, and other losses such as sorbtion. The tests would also provide data for

Figure 2-1. Study area, Ottawa River, Lima, Ohio.


Auglaize River
9
Kalida
Plum Creek
Columbus Grove
8
Rimer
Putnam Co.
Allen Co.
Sugar Creek
Pike Run
Ottawa River
Dug Run
Elida
Allentown
7
Lima
1
2
3
4
5
6
Ottawa River
Hog Creek
Little Hog Creek
Lafayette
Ada
Allen Co.
Hardin Co.
Grass Creek
STP
Refinery
Chemical Plant
Miles
0 1 2 3 4 5
Kilometers
0 2 4 6 8
N

Sampling Sites	River Distance (km)
Station 1	RK 74.0
Station 2	RK 60.7
Station 3	RK 60.3
Station 3A	RK 60.2
Station 3B	RK 59.7
Station 4	RK 59.4
Station 4A	RK 58.4
Station 5	RK 57.1
Station 6	RK 52.5
Station 7	RK 46.3
Station 8	RK 25.7
Station 8A	RK 12.9
Station 9	RK 1.6

prediction of ecological impact for comparison with the stream biological survey. Only *Ceriodaphnia* were used in the ambient toxicity test.

In the 1983 testing, each effluent was diluted with water collected above all known point source discharges because it was considered to be of good quality. This contrasts with the 1982 work in which diluent water came from upstream of each outfall. Also, unchlorinated STP effluent was not tested. Ambient stream toxicity tests were also performed in 1983; however, they included both *Ceriodaphnia* and fathead minnow larvae.

2.2 Field Survey Study Design

The study components of the 1982 field survey included quantitative assessment of the periphytic, benthic macroinvertebrate, and fish communities and a fish caging study. Nighttime drift samples at selected transects were collected to examine dispersal characteristics of the benthos and population structure of the macrozooplankton community.

The periphyton study investigated effects and recovery of the periphytic community by measuring chlorophyll *a* and biomass and by estimating composition and relative abundance. The relatively short reproduction time and rapid seasonal fluctuation in growth of periphytic algae make that community a useful indicator of localized effects resulting from effluent toxicity.

The benthic survey investigated ambient community response to the effluent(s) and approximated the location of downstream recovery should effluent discharge effects be substantiated. The benthic community is considered a good indicator of ambient response to adverse conditions because of its general lack of extensive mobility. The degree of community stability within affected areas can be measured by comparing composition and dominance to that of nonaffected areas.

In addition to benthic collections, drift collections were taken to evaluate the extent of colonizing potential from the source population, to measure the populations' response to plume entrainment, and to assess the resiliency of the drift dispersal mechanism immediately below the area of estimated influence.

The fisheries survey investigated the fish community to discern any changes in composition and dominance from previous surveys and to evaluate the response to the respective effluents.

In conjunction with the fisheries surveys, *in situ* fish caging surveys were performed to investigate *in situ* response to toxicity of resident fish species.

A hydrological analysis was conducted which consisted primarily of a dye study performed at each of three sites to identify the individual dilution characteristics of each effluent. By modeling downstream dilution contours for each discharger, the exposure concentration pertinent to instream effects could then be quantified. Ancillary flow measurements were also taken to estimate the flow contribution of the discharges to the receiving waterbody.

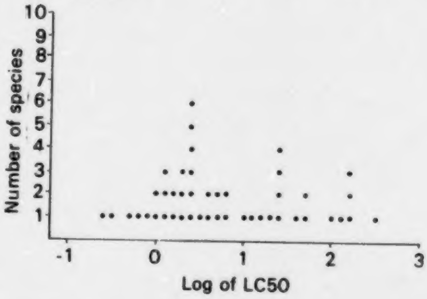
In the 1983 effort, a 2-day field survey was conducted in which the zooplankton, benthos, and fish were sampled at most of the same stations as in 1982. No hydrologic work was done during the 1983 survey.

2.3 Approach to Integretation of Laboratory and Field Efforts

Laboratory toxicity tests were conducted to measure the no-effect concentrations of three effluents. The toxicity of receiving water was also measured at various stream stations to confirm that the concentrations of effluents found to have effects in toxicity tests were similar to receiving water effluent concentrations that caused toxicity in the receiving stream. These data can then be compared to measures of the stream community data to validate or refute impacts predicted by the laboratory tests. If predictions were valid, then the effluent dilution test data could be used to estimate how much toxicity reduction would be necessary to diminish the impact.

Species sensitivity is one of the major problems in attempting to relate single species toxicity tests to impacted biological communites. One must have a degree of confidence that the test species is as sensitive as most of the community it is protecting. As an example, Figure 2-2 illustrates the results of copper LC50's for a number of species of freshwater organisms. Some species are sensitive at very low concentrations while other species survive concentrations that are several orders of magnitude higher. It is important to test a number of organisms to find a range of response levels in different discharge situations that, together, adequately predict instream community response. The purpose of measuring this range is to find the lowest end of the range of sensitivities, i.e., to find the more sensitive individual species. Toxicity to this species will represent the "worst-case" exposure conditions which will occur. Testing with a vertebrate (fish) and an invertebrate (*Ceriodaphnia*) provides a measure of toxicity that is not achieved by studying only one type of organism.

Figure 2-2. Histogram of LC50s for copper of fresh water species.



There is a high probability that the sensitivity of the species tested in chronic toxicity tests falls within the range of sensitivity of the species in the receiving stream, but usually the position of the species within the range is unknown. The sensitivity of the species tested will change depending on the types of toxicants.

The measures used in the toxicity tests, i.e., growth, reproduction, and mortality, are not the same measures of impact used for the biological assessment of the communities. As discussed above, the basic assumption is that the tested species are within the sensitivity range of the community, but it is not known where in that range they fall; therefore, the best way to compare the field and laboratory data would be to use measures of community response that include all species. If all species are included, then it does not matter whether the test species are in the most sensitive or least sensitive end of the range because some or many species in the community should show impact at the same or lower effluent concentrations. The response would be that their populations would be reduced in numbers from reference populations. If the affected species happen to be at lower trophic levels, there may also be impacts on other species which are not directly affected by toxicity, but are indirectly affected by alteration of the food chain.

3. Site Description

Several publicly owned treatment works (POTWs) and privately owned industries discharge treated effluents to the Ottawa River (Figure 2-1). The principle dischargers to which this study was directed are a sewage treatment plant (STP), a refinery, and a chemical company. The STP contributes the largest volume with a nominal flow of 0.53 m^3/sec (12 mgd), accounting for approximately 83 percent of the Ottawa River flow during the study period. As of the study period, the refinery discharges an average of 0.24 m^3/sec (5.5 mgd) and the chemical plant discharges about 0.088 m^3/sec (2.0 mgd).

The most upstream discharge is the STP. It is an activated sludge plant with ammonia removal and chlorination to provide a residual of 0.1-0.2 mg/liter of total chlorine. The plant appeared to be well operated and treatment efficiency was extremely good.

The next discharge is from a refinery located approximately 0.8 km downstream from the STP. The treatment plant had consisted of aerated lagoons with an approximate 15-day retention time during the 1982 study. Additional treatment was being brought online at the time of the 1983 study. This plant also treats wastewater from the agricultural chemical division of an adjacent chemical plant.

The third discharge is from a chemical plant. That discharge contained only cooling water at the time of the 1982 study because the industrial chemicals division was closed and the waste from the agricultural chemical division went to the refinery treatment plant. The industrial chemical division had been closed since the September 1982 study but when it was operating, ammonia was one of the main components of the waste. In 1983, a dry ice operation had been added and the effluent also contained discharge from the new agriculture section.

The study area on the Ottawa River in northwestern Ohio extended from Lima to Kalida, Ohio. It incorporated nearly 72 river kilometers (RK), 9 biological sampling locations, and 13 stations for ambient toxicity tests. The Ottawa River is about 15-20 m wide and has depths varying from 0.3 m in riffle areas to 1.5-2 m in pool areas. The primary habitat type is pools with infrequent riffle areas.

The station descriptions and approximate locations depicted in Figure 2-1 are:

1. Reference transect (RK 74.0) located above any influence from Lima at Thayer Road bridge.
2. Reference transect (RK 60.7) about 0.2 km above the STP, but downstream from other discharge points in the City of Lima.
3. Transect (RK 60.3) about 0.3 km below the discharge of the STP.

3a. Transect (RK 60.2) about 0.5 km below the discharge of the STP.

3b. Transect (RK 59.7) about 0.2 km above the refinery discharge.

4. Transect (RK 59.4) about 0.5 km below the discharge of the refinery.

4a. Transect (RK 58.4) 0.8 km below the chemical plant discharge.

5. Transect (RK 57.1) at Shawnee Road bridge 2.1 km below the discharge of the chemical plant.
6. Transect (RK 52.5) at Rt. 117 bridge just above the Shawnee STP.
7. Transect (RK 46.3) at Allentown Road bridge just below Shawnee STP and Allentown Dam.
8. Transect (RK 25.7) at Rimer at Rt. 198.

8a. Transect (RK 12.9) between Rimer and Kalida.

9. Transect (RK 1.6) below Kalida prior to the confluence with the Auglaize River.

All of the above stations were used for ambient water collections for the toxicity testing in 1982. Biological collections were made in 1982 at the above stations with the exception of those labeled "a" or "b." Stations 2, 3, 4, 5, 6, 7, and 8 were sampled in 1983 for the biological survey and ambient water collections.

Water temperature, dissolved oxygen (DO), specific conductance, and pH were measured during biological collections at each station in 1982. A Hydrolab Model 4041 was used for all measurements.

Water temperature ranged from 13.3 to 20.0°C during the study week in September 1982. The highest temperature occurred at stations near the plant outfalls (Stations 2 through 6). Conductivity ranged from 983 to 1,593 µmhos/cm. Discharges from the City of Lima, the refinery, and the chemical company all appeared to increase the conductivity, whereas the STP discharge decreased it. The pH values ranged from 6.2 to 8.4 throughout the study area.

Dissolved oxygen ranged from 3.4 mg/liter (35 percent saturation) at Station 7 to 15.9 mg/liter (greater than 150 percent saturation) at Station 8. Supersaturated conditions were found at the STP and refinery discharge stations, but these dropped to lower levels at the next three stations. DO measurements at Station 8 (taken at mid-afternoon), however, approached supersaturated levels. Although diel DO measurements were not made during this study, comparison between levels recorded at different times of day over the study period indicate that daily fluctuation at some stations was great. This was most pronounced at Stations 2, 5, and 7. However, DO levels equaling 75 percent saturation were recorded at the reference station (Station 2) at 0930 hours on 21 September and at 1800 hours on 24 September, indicating more stable water quality above the influence of the three discharges.

Stream flow was very low during both sample collection periods (September 1982 and July 1983). The flow prior to the STP discharge (station 2) was measured to be 0.11 m^3/sec in September 1982. The total flow at Kalida (Station 9) during the 1982 survey was estimated to be 1.08 m^3/sec.

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4. *Toxicity of Effluents and Receiving Water, 1982*

Toxicity tests were performed on each of the three effluents to measure subchronic effects on growth of larval fathead minnows and chronic reproductive effects on *Ceriodaphnia*. A range of effluent concentrations were used so acute mortality (if it existed) could be measured in addition to chronic toxicity.

The objective was to estimate the minimum concentration of each effluent that would cause acute mortality and chronic effects on growth (fathead minnows) or reproduction (*Ceriodaphnia*). These effect levels would then be compared to the effluent concentrations in the Ottawa River to predict where impact on resident species should occur. The validity of these predictions could be determined by an examination of the biotic condition of the stream at the locations where such effluent concentrations occurred as determined by the simultaneous stream biological survey and hydrological studies. The methods used for toxicity testing are described in Appendix A and follow those developed by Mount and Norberg (1984) and Norberg and Mount (in press).

4.1 Chemical/Physical Conditions

Temperatures were maintained between 22 and 25°C for the duration of the tests. The mobile laboratory did not have precise temperature control, but diel temperature changes were gradual and posed no problem.

The water hardness varied from 350 to 550 mg/liter as $CaCO_3$, depending on river station and effluent concentration. Alkalinity varied from 250 to 300 mg/liter. Only Station 1 had a hardness over 450 mg/liter. There were limestone quarries operating near Station 1 and dust from the operation may have caused a local increase in hardness. There were no surface streams draining from the quarries during the period of study.

Dissolved oxygen (DO) (mg/liter) was measured frequently because these were renewal tests and high biochemical oxygen demand (BOD) levels could cause DO to be depressed. The sewage treatment plant (STP) effluent had almost no measurable BOD, and, because the chemical plant effluent was all cooling water, BOD was probably similar. DO concentrations in the refinery waste were high during daylight hours as a result of dense algal populations. At night in the test chamber, especially in the fathead minnow tests, DO dropped markedly, due perhaps to a combination of algal respiration and BOD from bacterial action.

Initial DOs (when tests were set up) for the *Ceriodaphnia* tests ranged from 7.5 to 8.4 except for the ambient test Stations 4 through 7 where, on cloudy days, they ranged from 4.5 to 5.0. Final DO values (taken just before test solutions were changed) for the *Ceriodaphnia* tests were all above 5.0 except for two measurements (3.8 and 4.0) in the refinery test and two measurements (4.1 and 4.9) at Stations 6 and 7 in the ambient test. A total of 80 percent of DO readings were above 6.0 mg/liter.

Initial DOs for the fathead minnow tests were the same as for the *Ceriodaphnia* tests. After the first 24-hour period, DO ranged from 3.1 to 4.5 mg/liter. Upon discovery of this condition, feeding level was reduced and extra care exercised in siphoning the dead brine shrimp. Final values subsequently were above 5.0 mg/liter with several exceptions down to 4.0 mg/liter in the refinery effluent dilutions.

River pH varied depending on sunshine and time of day. In the tests, pH usually ranged from 8.0 to 8.2, both initial and final. The pH changed little during the 24-hour period of exposure to each sample. Inorganic suspended solids in the stream were very low because there was no runoff. The water color was greenish from algal blooms occurring in the pools above the dams in the City of Lima.

4.2 Results of Fathead Minnow Growth Test

Results of the testing of fathead minnow larvae exposed to various concentrations of the three effluents indicated that the STP had no effect on survival at any concentration (Table 4-1). The refinery waste caused substantial mortality at 50 percent effluent (Table 4-1, Figure 4-1) and a statistically significant (P = 0.05) amount of mortality at 10 percent effluent. The chemical plant waste was not toxic at 100 percent concentration, but toxicity was observed at the lower concentrations and also in the dilution water as shown by the low control survival[a]

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[a] Because the water used for controls and effluent dilution exhibited toxicity, a direct evaluation of the toxicity of the chemical plant is not possible

Table 4-1. **Seven-Day Percent Survival of Larval Fathead Minnows Exposed to Various Concentrations of Three Effluents in Upstream Water, Lima, Ohio, 1982**

Effluent by Replicate	% Effluent (v/v)					
	100	50	10	5	1	Control
STP						
A	100	90	100	100	80	100
B	100	100	90	100	100	90
C	100	100	100	100	80	100
D	100	100	100	90	100	100
Mean	100	97.5	97.5	97.5	90	97.5
Refinery						
A	0	20	70	80	100	90
B	0	30	80	100	90	90
C	0	0	80	80	100	100
D	0	0	70	100	100	100
Mean	0	12.5	75	90	97.5	95
Chemical Plant						
A	80	70	40	40	50	40[a]
B	90	70	40	20	50	20
C	100	80	50	40	60	40
D	90	60	40	36.4	50	20
Mean	90	70	45	34.1	52.5	33.3

[a] Controls reflect presence of refinery waste as do the lower effluent concentrations of the chemical plant.

(Table 4-1). Based on the results of the hydrological measurements taken concurrently, the dilution water for the chemical plant discharge (taken below the refinery discharge point) was approximately 29 percent refinery waste (Table 4-2), which is a concentration high enough to account for the mortality observed in the chemical plant waste control and the 1, 5, and 10 percent concentrations. The 10 percent chemical plant waste dilution was estimated to be 24 percent refinery waste.

The final fathead minnow weights after 7 days' exposure to the three effluents diluted with upstream water are given in Table 4-3 and Figure 4-1. The relative toxicity as based on growth is essentially identical to the effect on mortality indicating that the toxicity of the refinery waste on fathead minnow growth occurred at about 10 percent effluent concentration. This toxicity of the refinery affected the results of the growth test performed on the chemical plant effluent.

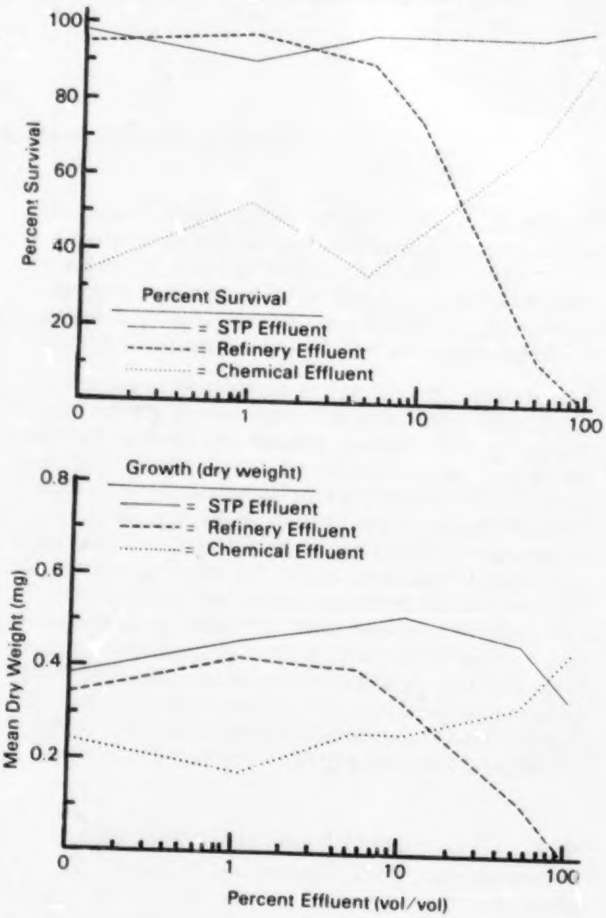
Two dilution waters (Lake Superior and the receiving water) were used in testing concentrations of the refinery effluent in a side-by-side comparison to ascertain the inherent toxicity of the refinery waste (Table 4-4). However, fathead minnow growth was similar in the two dilution waters with slightly higher weights attained in the refinery waste concentrations using the Lake Superior dilution water (Table 4-5, Figure 4-2). No statistical analysis was performed to test this difference.

Table 4-2. **Average Concentration (%) of Stream Water and Effluent Below Each Discharge During the 1982 Testing Period[a]**

Location	Concentration of Effluent (%)			
	STP	Refinery	Chemical Plant	Stream Water
Below STP	77	0	0	23.0
Below refinery	57.7	28.8	0	13.5
Below chemical plant	52.5	26.9	9	11.6

[a] Based on hydrological measurements expanded in Table 6-3.

Figure 4-1. **Fathead minnow growth tests for STP, refinery and chemical effluents, Lima, Ohio, 1982.**



4.3 Results of Reproductive Potential Tests Using *Ceriodaphnia*

The young production of *Ceriodaphnia* in various effluents diluted with stream water immediately upstream from each outfall was the primary focus of the reproductive potential tests. The mean young/female is calculated as the total young produced in 7 days at each concentration divided by the original number (10) of animals used. Therefore, early mortality of the original animals will reduce the young per

Table 4-3. **Mean Dry Weight (mg) of Larval Fathead Minnows Exposed to Three Effluents at Various Concentrations, Lima, Ohio, 1982**

Effluent by Replicate	% Effluent (v v) 100	50	10	5	1	Control
STP						
A	0.32	0.46	0.57	0.47	0.52	0.41
B	0.32	0.51	0.54	0.52	0.42	0.37
C	0.36	0.40	0.51	0.56	0.49	0.45
D	0.35	0.47	0.51	0.45	0.43	0.29
Mean	0.34	0.46	0.53	0.50	0.46	0.38
SD	0.02	0.04	0.03	0.05	0.05	0.07
Refinery						
A	--	0.06	0.39	0.40	0.42	0.31
B	--	0.18	0.36	0.42	0.43	0.36
C	--	--	0.28	0.37	0.46	0.30
D	--	--	0.29	0.42	0.37	0.41
Mean	--	0.12	0.33	0.40	0.42	0.34
SD	--	0.09	0.05	0.02	0.04	0.05
Chemical Plant						
A	0.43	0.29	0.33	0.19	0.26	0.23[a]
B	0.39	0.35	0.23	0.30	0.19	0.13
C	0.36	0.36	0.28	0.25	0.09	0.22
D	0.58	0.30	0.20	0.28	0.14	0.25
Mean	0.44	0.32	0.26	0.26	0.17	0.21
SD	0.10	0.04	0.06	0.05	0.07	0.06

[a] Controls for chemical plant were affected by toxicity of refinery waste.

female regardless of how rapid young production of surviving adults may be. The percent survival of the test organisms was also compared among test concentrations to measure lethality.

The 100 and 50 percent chlorinated STP effluent caused mortality in 2-5 days (Table 4-6). The mortalities in the 10, 5, and 1 percent concentrations and controls were thought to be caused by the occurrence of a toxic slug of water from upstream, as described later in this section. The existence of a toxic slug of water from an upstream source is hypothesized on the basis of mortality of control organisms and test organisms not attributable to known effluents. Because of this mortality, the young per female is lower than that which would have been obtained if they had survived to 7 days.

Table 4-4. **Seven-Day Percent Survival of Larval Fathead Minnows Exposed to Refinery Waste Diluted with Two Different Dilution Waters, Lima, Ohio, 1982**

Replicate	% Effluent (v/v) 100[a]	50	10	5	1	Control
Receiving Water[b]						
A	0	20	70	80	100	90
B	0	30	80	100	90	90
C	0	0	80	80	100	100
D	0	0	70	100	100	100
Mean	0	12.5	75	90	97.5	95
Lake Superior Water						
A		10	90	100	60	50
B		10	80	80	60	60
C		0	100	100	60	90
D		0	90	100	70	90
Mean		10	90	95	62.5	72.5

[a] 100 percent effluent--no dilution water used; only four replicates tested.

[b] Dilution water taken immediately upstream of the refinery discharge.

Table 4-5. **Mean Dry Weight (mg) of Larval Fathead Minnows Exposed to Refinery Effluents Diluted with Two Different Dilution Waters, Lima, Ohio, 1982**

Replicate	% Effluent (v v) 100[a]	50	10	5	1	Control
Receiving Water[b]						
A	--	0.06	0.39	0.40	0.42	0.31
B	--	0.18	0.36	0.42	0.43	0.36
C	--	--	0.28	0.37	0.46	0.30
D	--	--	0.29	0.42	0.37	0.40
Mean		0.12	0.33	0.40	0.42	0.34
SD		0.09	0.05	0.02	0.04	0.05
Lake Superior Water						
A		0.24	0.32	0.38	0.40	0.34
B		0.10	0.45	0.41	0.42	0.49
C		--	0.40	0.42	0.35	0.40
D		--	0.41	0.41	0.31	0.38
Mean		0.17	0.40	0.40	0.37	0.40
SD		0.10	0.05	0.07	0.05	0.06

[a] 100 percent effluent--no dilution water used; only four replicates tested.

[b] Dilution water taken immediately upstream of the refinery discharge.

The refinery waste was lethal at 100 percent and nonlethal at 50 percent. The mortalities in 10, 5, and 1 percent waste and control were due to the STP effluent in the dilution water. The dilution water used for the refinery waste was 77 percent STP effluent (Table 4-2). In the 10 percent refinery concentration, there was an approximate 69 percent concentration of STP effluent—clearly enough to be lethal as indicated from the STP test. Even at the 50 percent refinery effluent concentration, there is 38 percent STP effluent, which is probably toxic.

Figure 4-2. **Comparison of fathead minnow weights in refinery effluent using two dilution waters, Lima, Ohio, 1982.**

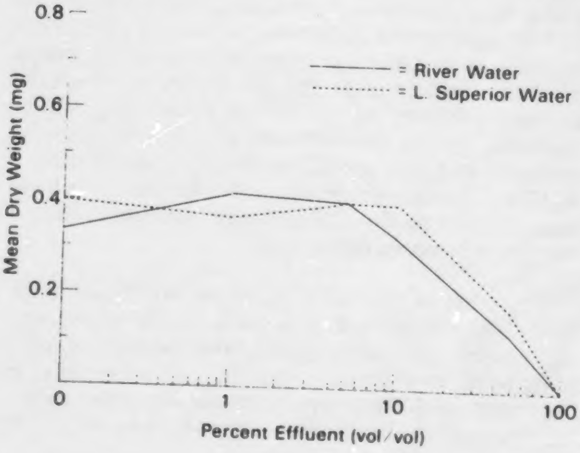
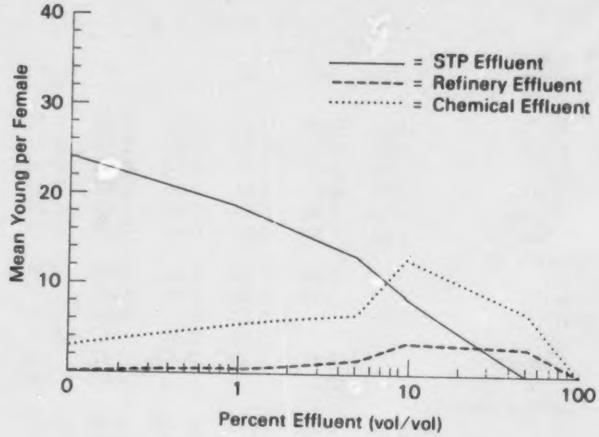


Table 4-6. **Mean Young Per Original Female and Mean Percent Survival of Adult *Ceriodaphnia* in Various Effluent Concentrations Using Receiving Water for Dilution, Lima, Ohio, 1982**

% Effluent (v/v)	Young/ Female ($\bar{x}$)	SD	Days 1	2	3	4	5	6	7
			Percent Survival						
		Cl_2 STP Effluent							
100	--	--	100	100	0	0	0	0	0
50	0.1	--	100	90	80	40	0	0	0
10	8.6	3.1	100	100	100	100	100	0	0
5	13.3	7.8	100	100	100	100	100	10	10
1	18.7	8.7	100	90	90	0	90	0	0
C	24.2	5.0	100	100	100	100	100	80	0
		Refinery Effluent							
100	0	--	90	0	0	0	0	0	0
50	3.2	2.8	100	90	90	90	90	90	90
10	3.5	4.2	100	100	100	100	0	0	0
5	1.7	1.9	100	100	100	80	0	0	0
1	0.4	--	100	100	80	30	0	0	0
C[a]	0	--	100	100	10	0	0	0	0
		Chemical Plant Effluent							
100	0.7	--	90	80	70	40	0	0	0
50	7.2	3.8	100	100	100	100	100	90	80
10	12.9	4.2	100	100	100	100	100	90	50
5	6.7	1.9	100	90	90	90	90	40	30
1	5.8	--	100	100	100	100	100	10	0
C[a]	3.3	--	100	100	100	100	90	40	10
		$UnCl_2$ STP Effluent							
100	0	--	100	80	0	0	0	0	0
50	0.9	--	100	90	60	20	0	0	0
10	20.5	5.7	100	100	100	100	100	100	90
5	0	--	100	100	100	100	0	0	0
1	0.1	--	100	90	60	10	10	10	10
C	25.8	4.6	100	100	100	100	100	100	100

[a] Controls for the refinery effluent test reflect presence of STP effluent, and in the chemical effluent test the combined toxicity of STP and refinery is seen.

Figure 4-3. ***Ceriodaphnia* young production in three effluents, Lima, Ohio, 1982.**



The chemical plant waste was lethal at 100 percent (Table 4-6). The mortality at concentrations equal to or less than 10 percent were caused by the toxic slug of water mentioned previously. The dilution water used for this test was about 58 percent STP effluent and at 50 percent chemical waste concentration, the STP effluent would have been at about a 29 percent concentration which again is high enough to produce mortality, except that the presence of about 14 percent refinery waste would have delayed the mortality due to the STP effluent. The progressive mortality and low young per female (7.2) on Days 6 and 7 at 50 percent chemical plant effluent may have been caused by the STP effluent, the toxic slug, or some combination of the two.

The interactive nature of the three effluents is illustrated in Figure 4-3 in which young production increases in the refinery and chemical plant effluent tests as concentration increases, or, conversely, as volume of the STP effluent in the test water decreases. However, young production also was reduced in 100 percent concentration of the refinery and chemical plant effluents which suggested chronic toxicity.

To understand the true toxicity of the refinery waste, the results of the test with refinery waste and Lake Superior water must be examined (Table 4-7). In that test, the 10 percent waste concentration was nontoxic and 50 percent had a significant (P = 0.05) effect on young production. When these results are compared to those using the receiving water for dilution, it can be concluded that the combination of refinery toxicity and STP toxicity in the dilution water caused adverse effects on young production at all concentrations. Further, the presence of small concentrations of refinery waste delayed the toxicity of the STP effluent. This is supported by the fact that 5 percent waste doubled the survival time with only a 4 percent lower concentration of STP effluent as compared to the control.

The test with unchlorinated STP waste and Station 1 water (Table 4-6) was conducted to determine if the toxicity in the STP effluent tests was caused by chlorination or something in Station 2 water which

Table 4-7. **Mean Young Per Original Female and Percent Survival of Adult *Ceriodaphnia* in Refinery Effluent Concentrations Using Lake Superior Water for Dilution, Lima, Ohio, 1982**

% Effluent (v/v)	Young/ Female ($\bar{x}$)	SD	Days 1	2	3	4	5	6	7
			Percent Survival						
100	0	--	100	0	0	0	0	0	0
50	6.9	5.3	100	100	100	100	100	100	100
10	21.6	8.0	100	100	100	100	90	90	90
5	25.0	6.1	100	100	100	100	100	100	100
1	17.1	9.6	100	100	90	90	90	90	90
C	15.9	9.8	90	90	90	90	90	90	80

might be attributable to runoff from the City of Lima. The young per female and survival of the control shows that Station 1 was acceptable. The very regular pattern of mortality at 100, 50, and 1 percent, but not 10 percent, and the comparable effect on young production does not seem likely due to chance, sick animals, or contamination in the laboratory. The mortality pattern in the 100 percent unchlorinated effluent is nearly identical to that for the chlorinated effluent so apparently the toxicant was not a product of chlorination. The unchlorinated controls give no evidence of the toxic slug shown by the chlorinated waste controls.

One possible explanation for this unusual dose response curve is that the toxicant was very pH-dependent and was in a more toxic form at the pH of these lower concentrations of waste. The effluent has a pH from 7.2 to 7.4 and the control water was 8.3 or higher. As the percent waste decreased, the pH increased, approaching that of the control.[a]

The results of the ambient test for persistence of toxicity are presented in Table 4-8 and Figure 4-4. At Station 3, 3A, and 3B, which contained 77 percent STP effluent but no refinery waste, the mortality started between Day 2 and 3 as would be predicted from Table 4-6 and the test on chlorinated effluent. Station 4 water contained 58 percent STP effluent and 29 percent refinery waste. Mortality was delayed by several days even though the drop in STP effluent was only from 77 to 58 percent STP between Stations 3B and 4. The increase in mortality between Day 5 and 6 at Stations 3B and 4 coincides with the mortality at Station 2 which had no STP or refinery effluent. Note that at Station 6, which is almost 5 km below Station 5, the increase in mortality was a day later, occurring between Day 6 and Day 7. These observations are well explained by the toxic slug hypothesis mentioned earlier.

The controls for the STP effluent dilution (ED) test on chlorinated effluents showed a sharp rise in mortality one day later than did the animals in the ambient test at Station 2. Yet, the test water for both the controls and the ambient Station 2 animals was taken from the same 5-gallon sample, with the only difference being that the sample sat overnight before being used for the ED test. It could be speculated that the difference was caused by the aging of the water used for the ED test; however, the sudden mortality increase in chlorinated STP concentrations of 1, 5, and 10 percent occurred on the same day as for the Station 2 animals. The sudden rise in mortality in the controls and 1 and 5 percent concentrations of the chemical plant ED test occurred also on the same day as the mortality in the ambient test.

Since the mortality occurred on the same day in three STP concentrations, in three concentrations in the chemical plant ED test, and in four ambient stations but on a different day for the chlorinated STP ED test controls, the most logical explanation seems to be that the control animals in the STP ED tests were a bit more resistant. There was, however, a 20 percent mortality in these controls the same day as the mortality which occurred at ambient Station 2.

Young production was much lower at Stations 3 and 5, and was much higher than any upstream stations at Stations 8A and 9. The high number of young at these latter stations undoubtedly reflects the nutrient enrichment from upstream and the increased food (in the form of bacteria) levels in the absence of toxic materials. There was very little input to the stream below the outfalls so the reduced toxicity was probably due to degradation and other loss factors rather than dilution.

The young production data at Stations 2 through 6 is affected by the toxic slug and therefore is not only the result of the effluents present.

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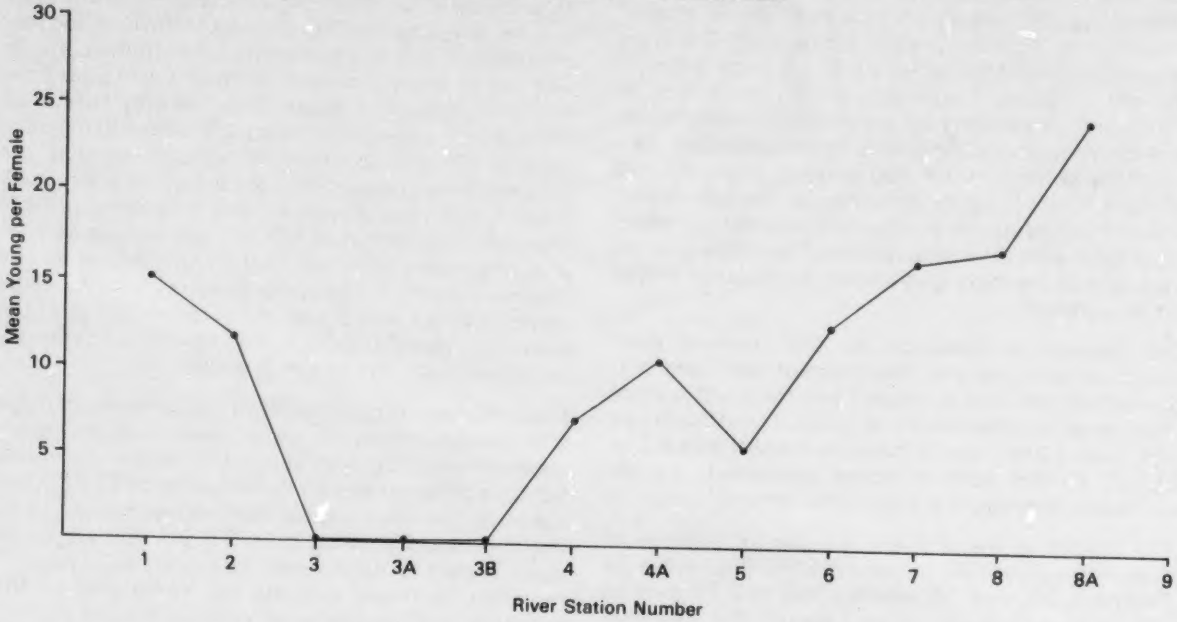
[a] Since this work was completed, similar toxicity has been observed in other STP effluents.

Table 4-8. Mean Young Production and Percent Survival of *Ceriodaphnia* for the Ambient Toxicity Tests in 1982

Station	Station Description	River Kilometer	Young/ Female (x̄)	SD	Final Survival (%)	Daily Survival (%)						
						1	2	3	4	5	6	7
1	Above Lima	74.0	15.5	8.0	90	100	100	100	90	90	90	90
2	Above STP	60.7	14.1	2.1	0	100	100	100	100	90	10	0
3	Below STP	60.3	0	--	0	100	100	10	0	0	0	0
3A	Midway between STP and refinery	60.2	0	--	0	100	100	10	0	0	0	0
3B	Above refinery	59.7	0.4	--	0	90	90	40	0	0	0	0
4	Above chemical plant	59.4	7.5	3.6	10	100	100	100	100	100	50	10
4A	Below chemical plant	58.4	11.1	4.6	30	100	100	100	100	100	40	30
5	Shawnee Bridge	57.1	5.7	4.0	0	90	90	90	90	90	60	0
6	Route 117	52.5	12.6	3.8	10	100	100	100	100	100	100	10
7	Allentown	46.3	16.8	6.1	100	100	100	100	100	100	100	100
8	Rimer	25.7	17.4	9.5	80	100	90	90	90	90	80	80
8A	"Boonie" Station	12.9	25.0	3.3	100	100	100	100	100	100	100	100
9	Kalida	1.6	25.6	5.5	100	100	100	100	100	100	100	100

Figure 4-4. *Ceriodaphnia* young production for ambient stations, Lima, Ohio, 1982.



4.4 Evaluation of Toxicity Impact[a]

The results of these laboratory tests showed that the concentrations of STP effluent that are toxic to *Ceriodaphnia* are exceeded below the STP outfall in the river, but that there is no toxicity to fathead minnows. The concentration of refinery effluent below that outfall is high enough to produce adverse impact on fathead minnows. The effect level of the refinery waste plus the STP waste in the dilution water is less than the concentrations existing in the stream below the refinery and therefore toxicity to *Ceriodaphnia* in the river would be expected. From the refinery/Lake Superior water tests, the no effect level lies between 10 and 50 percent. The concentration of refinery effluent in the river below the outfall is 29 percent. To determine if refinery waste alone would be toxic to *Ceriodaphnia* below the outfall in the river if no STP effluent were present, tests of refinery waste without STP effluent and closer spacing of effluent concentrations would be needed.

The toxicity contributed by the chemical plant effluent in the river cannot be assessed without other tests. There is enough refinery effluent in the dilution water to cause adverse effects on the fathead minnows and there is sufficient STP effluent in the dilution water to cause adverse effects on the *Ceriodaphnia*.

Tests that measure the inherent toxicity of each effluent without the complication of other effluents being present are needed if the contribution from each effluent is to be assessed. If a prediction of the impact after each discharge is required, however, the tests must be done with the extant concentrations of the effluent in the stream below the outfall present.

The symptoms caused by the STP and refinery effluent were quite different for *Ceriodaphnia* and the effluent causing the dominant toxicity could be recognized by the symptomology. No toxicity of STP and chemical plant effluents to fathead minnows was found, so all toxicity to them could be ascribed to the refinery waste.

The dilution studies to determine the effluent concentrations in the river are a very necessary part of the assessment. Without them, little interpretation could be done. During this study, flow was low and stable and had been so for some time previous to the study. Variable flows before and during the study would make predictions very difficult, not because the tests are invalid, but because the actual ambient exposure the animals receive would not be known. In those cases where exposure varies greatly, either as a result of variable effluent quality or variable stream flow, laboratory tests would have to be conducted so as to better mimic ambient concentration variations.

The absence of waste from the industrial chemical division of the chemical plant introduces another unknown in predicting the stream impact. The biological survey may have measured impacts from earlier exposure to this component if residual effects still existed. This could cause some discrepancy between the lab tests done in this study and the associated field data.

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[a] Because control and dilution water was toxic in some tests, the performance of control animals cannot be used as a basis for comparison to test concentration in all cases.

The toxicity test results are confounded by the toxic slug of water from an upstream source. This study approach was designed to measure chronic effects and not episodic events. Further, there is no way to know how frequently such spills occur, and therefore, no way to know how much they affect the instream biota. Tests could be designed to assess such cases.

Within the limits of this study, the ambient toxicity data indicate that the conditions in the river reach where the discharges occur are adverse to *Ceriodaphnia* and that the conditions of Station 8A and 9 are not. Because dilution water inputs are very minor downstream of the discharge, one can conclude that the loss of ambient toxicity is due to factors other than dilution. Visual observation of the dye additions indicate a time-of-travel of a week or more from the STP outfall to Station 9. This at least puts some bounds on the half-life of the combined toxicity of the three effluents.

The test data from the ED tests for the three effluents and the two species show the importance of differences in sensitivity of various animal species and the need to factor this into toxicity measurements. If only the fathead minnows had been tested, the toxic component in the STP to *Ceriodaphnia* would have been missed completely. Yet, that toxicity may impact other types of invertebrates substantially in the stream.

5. *Toxicity of Effluents and Receiving Water, 1983*

On 7-8 July 1983, the Ottawa River was revisited for a brief survey after operational modifications were made to the refinery. On 8 July 1983, a 24-hour composite sample from each of the three effluents and grab samples from seven stream stations were taken. These samples along with a large volume of water from Station 1, were iced and transported back to Duluth. Testing began on 10 July 1983 in a mobile trailer located at the laboratory. Changes in methodology from 1982 to 1983 were:

1. A single 24-hour composite sample was used for the entire test but solutions were renewed each day.
2. A fathead minnow ambient toxicity test was run on the stream station samples in addition to the *Ceriodaphnia* test.
3. Each effluent was diluted with Station 1 water only.
4. No unchlorinated STP effluent was tested.

The methods used in these tests are more fully described in Appendix A.

5.1 Results

Tables 5-1, 5-2, and 5-3 contain the results of the routine water chemistry. All dissolved oxygen (DO) measurements were above 5 mg/liter. The pH was similar for all exposures and the conductivity varied depending on the effluent and the amount of dilution.

Table 5-2. **Water Chemistry Data for Ambient Toxicity Test with Fathead Minnows at Various River Stations, Lima, Ohio, 1983**

Ambient Station	pH Range	Initial DO (mg/l) $\bar{x}$	Initial DO (mg/l) SD	Final DO (mg/l) $\bar{x}$	Final DO (mg/l) SD	Conductivity (μmhos/cm)
Station 2[a]	7.9-8.1	8.6	(0.4)	8.0	(1.0)	790
Station 3	7.9-8.0	8.6	(0.4)	7.4	(0.5)	830
Station 4	7.7-7.8	8.7	(0.5)	7.9	(0.7)	1,480
Station 5	7.7-7.9	8.5	(0.4)	8.5	(1.0)	1,350
Station 6	7.8-7.9	8.5	(0.3)	8.5	(0.9)	1,210
Station 7	7.8-7.9	8.6	(0.2)	8.3	(0.9)	1,220
Station 8	7.8-8.0	8.7	(0.4)	7.5	(0.4)	1,050

[a] For Station 1 see controls on Table 5-1.

The data for fathead minnow 7-day survival and weight for the three effluents diluted with Station 1 water are presented in Tables 5-4 and 5-5 and Figure 5-1. There was no measurable effect of any concentration of sewage treatment plant (STP) effluent on either survival or growth. The no-effect concentration for the refinery waste was between 100 and 30 percent. The growth observed in all concentrations of the chemical plant effluent were significantly different ($P = 0.01$) from the controls, but there was not a typical dose-response curve.

Table 5-1. **Chemistry Data for Three Effluents in Station 1 Water for Fatheat Minnow Larval Growth Tests, Lima, Ohio, 1983**

Effluent	Percent Effluent (v/v)	pH Range	Initial DO (mg/l) $\bar{x}$	Initial DO (mg/l) SD	Final DO (mg/l) $\bar{x}$	Final DO (mg/l) SD	Conductivity (μmhos/cm)
STP	100	7.7-7.9	8.6	(0.2)	6.4	(0.3)	850
	30	7.7-7.8	8.6	(0.2)	6.8	(0.5)	750
	10	7.6-7.8	8.6	(0.1)	6.8	(0.5)	710
	3	7.6-7.7	8.3	(1.0)	7.1	(0.3)	700
	1	7.5-7.8	8.2	(1.0)	7.2	(0.3)	690
	C[a]	7.5-7.9	8.7	(1.5)	7.1	(0.5)	690
Refinery	100	7.2-7.5	8.4	(0.8)	7.3	(0.2)	2,700
	30	7.6-7.8	8.6	(0.5)	7.0	(0.3)	1,280
	10	7.7-7.8	8.6	(0.5)	7.0	(0.3)	900
	3	7.7-7.9	8.7	(0.4)	6.7	(0.7)	750
	1	7.9	8.7	(0.4)	6.9	(0.5)	690
	C	8.0	8.5	(0.4)	6.8	(0.5)	690
Chemical plant	100	7.8	8.5	(0.5)	7.6	(0.7)	2,050
	30	7.9	8.5	(0.7)	7.2	(0.7)	1,100
	10	7.9-8.0	8.5	(0.4)	7.2	(0.4)	820
	3	7.9-8.0	8.5	(0.4)	7.1	(0.3)	720
	1	7.9-8.0	8.5	(0.4)	6.9	(0.4)	710

[a] Water for controls and dilution was taken from Station 1 upstream of all discharges.

Table 5-3. Final Dissolved Oxygen Concentrations for *Ceriodaphnia* Tests on Effluents and Stream Station Water, Lima, Ohio, 1983

Sample	% Effluent (v/v)	DO (mg/l) $\bar{x}$	DO (mg/l) SD
STP	100	8.0	(0.7)
	30	7.7	(0.5)
	10	7.7	(0.5)
	3	7.8	(0.5)
	1	7.8	(0.5)
	C	7.7	(0.5)
Refinery	100	7.2	(0.4)
	30	7.4	(0.2)
	10	7.6	(0.3)
	3	7.5	(0.2)
	1	7.5	(0.2)
	C	7.4	(0.2)
Chemical plant	100	7.3	(0.3)
	30	7.5	(0.3)
	10	7.4	(0.1)
	3	7.4	(0.1)
	1	7.5	(0.1)
	C	7.4	(0.2)
Station 2		7.4	(0.5)
Station 3		7.4	(0.5)
Station 4		7.4	(0.4)
Station 5		7.4	(0.3)
Station 6		7.4	(0.4)
Station 7		7.4	(0.3)
Station 8		7.3	(0.3)

The data for the ambient toxicity to fathead minnows show that the weights at Stations 2 and 8 were all significantly lower (P = 0.01) than the fish in Station 1 water but not for percent survival (Tables 5-6 and 5-7 and Figure 5-2).

The data for the *Ceriodaphnia* young production and survival for the three effluents diluted in water from Station 1 are shown in Table 5-8 and Figure 5-3. The no-effect level was between 10 and 3 percent for both the refinery and the STP while no concentration of the chemical plant produced an effect.

Table 5-4. Seven-Day Percent Survival of Larval Fathead Minnows Exposed to Various Concentrations of Three Effluents in Station 1 Water, Lima, Ohio, 1983

Effluent by Replicate	% Effluent (v/v) 100	30	10	3	1	Control
STP						
A	70	100	100	100	90	90
B	80	90	100	90	90	90
C	100	90	90	90	100	90
D	90	100	80	70	90	90
Mean	85	95	92.5	87.5	92.5	90[a]
Refinery						
A	20	100	100	100	100	100
B	10	90	100	90	90	80
C	0	100	100	90	90	100
D	20	90	90	100	90	80
Mean	16.6[b]	95	97.5	95	92.5	90[a]
Chemical plant						
A	80	100	80	80	70	—
B	100	80	90	70	70	—
C	70	60	70	90	90	—
D	100	70	90	80	80	—
Mean	87.5	77.5	82.5	80	77.5	90

[a] Data for two controls pooled.
[b] Mean of treatments significantly different (P = 0.01) from control.

Figure 5-1. Fathead minnow growth tests for STP, refinery, and chemical effluents, Lima, Ohio, 1983.

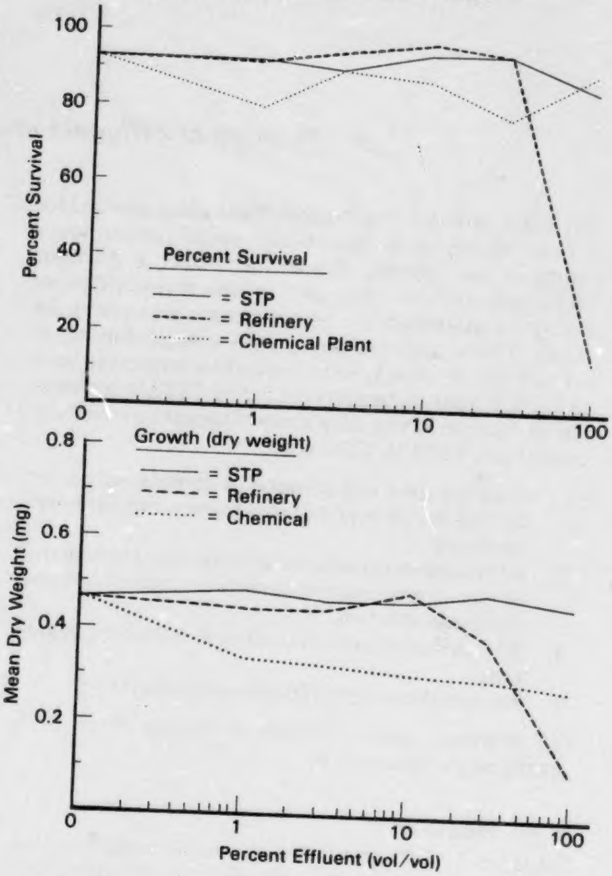


Table 5-9 and Figure 5-4 contain data for *Ceriodaphnia* survival and young production for the ambient toxicity test. Stations 3 through 7 were significantly different (P = 0.01) from Station 2 but Station 8 was not different.

The data for the estimated effect concentrations for the three effluents and two species are given in Table 5-10. Table 5-11 presents the estimated concentrations of STP and refinery effluent in the Ottawa River below each outfall based on conductivity. Conductivity at Station 5 was less than the conductivity at Station 4 and, therefore, the dilution of the chemical plant effluent cannot be calculated. In 1982, the chemical plant made up <10 percent of the river and conditions were similar in 1983.

Table 5-5. **Mean Weight (mg) of Larval Fathead Minnows Exposed to Three Effluents at Various Concentrations, Lima, Ohio, 1983**

Effluent by Replicate	% Effluent (v/v) 100	30	10	3	1	Control
STP						
A	0.450	0.400	0.445	0.410	0.522	0.511
B	0.450	0.567	0.475	0.539	0.498	0.478
C	0.470	0.483	0.470	0.478	0.490	0.500
D	0.483	0.520	0.494	0.443	0.456	0.544
Mean	0.46	0.49	0.47	0.47	0.49	0.47[a]
SD	(0.02)	(0.07)	(0.02)	(0.06)	(0.03)	(0.03)
Refinery						
A	0.075	0.410	0.560	0.405	0.470	0.435
B	0.050	0.411	0.440	0.461	0.472	0.386
C	—	0.345	0.545	0.511	0.378	0.435
D	0.175	0.411	0.406	0.425	0.472	0.444
Mean	0.1[b]	0.39	0.49	0.45	0.45	0.47[a]
SD	(0.07)	(0.03)	(0.08)	(0.05)	(0.05)	(0.03)
Chemical plant						
A	0.281	0.310	0.338	0.275	0.286	—
B	0.310	0.325	0.294	0.343	0.364	—
C	0.279	0.275	0.314	0.333	0.317	—
D	0.265	0.271	0.275	0.363	0.412	—
Mean	0.28[b]	0.30[b]	0.31[b]	0.33[b]	0.34[b]	0.47[a]
SD	(0.02)	(0.03)	(0.03)	(0.04)	(0.06)	(0.03)

[a] Data for two controls pooled.
[b] Mean of treatments significantly different (P = 0.01) from control.

Table 5-6. **Seven-Day Percent Survival of Larval Fathead Minnows Exposed to Water from Various Stream Stations for Ambient Toxicity, Lima, Ohio, 1983**

Replicate	Stream Station 1[a]	2	3	4	5	6	7	8
A		100	80	90	80	90	100	90
B		100	100	90	90	90	80	90
C		80	90	60	80	90	90	80
D		100	70	70	100	80	90	90
Mean	90	95	85	77.5	87.5	87.5	90	87.5

[a] Data for Station 1 are the same as for controls on Table 5-4.

5.2 Discussion

Concentrations of STP and refinery waste in the stream were less than the minimum predicted effect levels for fathead minnows (Tables 5-10 and 5-11). The ambient toxicity data as depicted by Figure 5-4 does not suggest an increase in toxicity below either the STP or refinery outfall (Stations 3 and 4) as compared to Station 2 which is above both outfalls. However, all three stations and Stations 5 through 8 were less than Station 1.

For *Ceriodaphnia*, both the STP and the refinery concentrations were greater in the river than the estimated effect concentrations, but the chemical plant concentrations was not. The ambient toxicity data in Table 5-8 show significant toxicity below the

Table 5-7. **Mean Weight (mg) of Larval Fathead Minnows Exposed to Water from Various Stream Stations for Ambient Toxicity, Lima, Ohio, 1983**

Replicate	Stream Station 1[a]	2	3	4	5	6	7	8
A		0.35	0.27	0.38	0.31	0.28	0.30	0.34
B		0.36	0.25	0.32	0.39	0.33	0.37	0.33
C		0.37	0.34	0.28	0.30	0.26	0.31	0.37
D		0.30	0.34	0.36	0.35	0.32	0.37	0.26
Mean	0.47	0.34[b]	0.30[b]	0.34[b]	0.34[b]	0.30[b]	0.34[b]	0.33[b]
SD	(0.03)	(0.03)	(0.05)	(0.04)	(0.04)	(0.03)	(0.04)	(0.05)

[a] Data for Station 1 are the same as for controls on Table 5-5.
[b] Significantly different (P = 0.01) from controls in effluent test (Station 1 water).

Figure 5-2. **Fathead minnow growth tests for ambient stations, Lima, Ohio, 1983.**

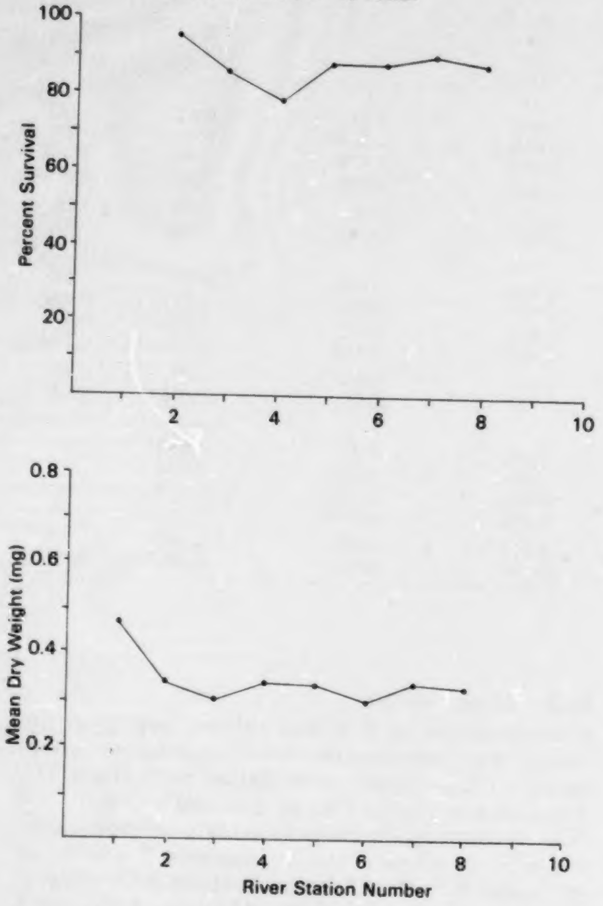


Figure 5-3. ***Ceriodaphnia* young production in three effluents, Lima, Ohio, 1983.**

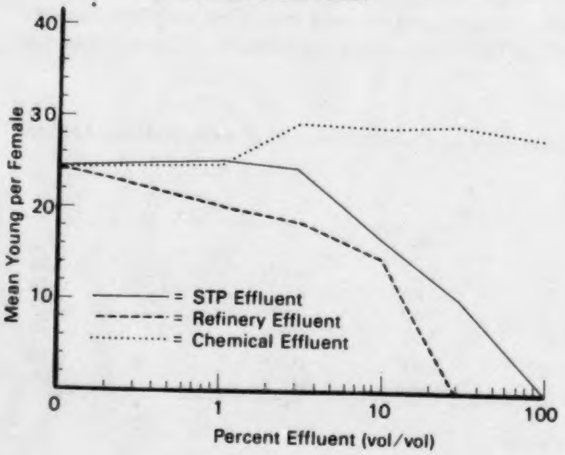


Table 5-8. **Mean Young Per Female *Ceriodaphnia* and Mean 7-Day Percent Survival of Original Test Animals Exposed to Various Effluent Dilutions in Station 1 Water, Lima, Ohio, 1983**

Effluent	% Effluent (v/v)	7-Day Survival (%)	Young/Female $\bar{x}$	SD
STP	100	0	0.30[b]	(0.95)
	30	30	10.6[b]	(4.5)
	10	90	17.0[b]	(10.0)
	3	100	24.5	(3.2)
	1	100	25.2	(2.2)
	Control[a]	93	24.6	(7.1)
Refinery	100	70	0.0[b]	—
	30	100	0.0[b]	—
	10	100	14.7[b]	(7.9)
	3	80	18.5	(11.0)
	1	80	20.2	(11.6)
	Control	93	24.6	(7.1)
Chemical plant	100	100	27.7	(7.4)
	30	100	29.1	(6.8)
	10	100	28.8	(3.2)
	3	100	29.2	(2.6)
	1	100	24.6	(3.1)
	Control	93	24.6	(7.1)

[a] Control data for all three combined.

[b] Significantly different (P = 0.01) from combined controls.

Table 5-9. **Mean Young Per Female *Ceriodaphnia* and 7-Day Percent Survival of Original Test Animals Exposed to Water from Various Stream Stations for Ambient Toxicity, Lima, Ohio, 1983**

Station No.	7-Day Survival (%)	Young/Female $\bar{x}$	SD
1	93[a]	24.6[a]	7.1
2	100	29.2	7.1
3	0	2.7[b]	2.5
4	90	11.9[b]	4.6
5	20	8.4[b]	5.3
6	100	4.2[b]	2.6
7	10	6.8[b]	5.6
8	100	27.9	3.3

[a] Average of three sets of effluent controls.

[b] Significantly different (P = 0.01) from Station 2.

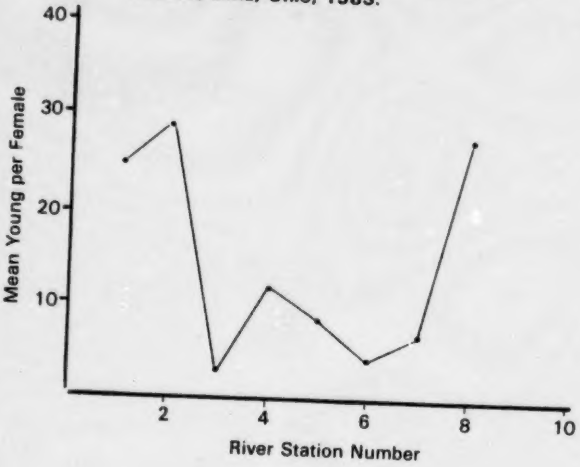
Table 5-10. **Geometric Mean of the Effect and No Effect Concentration for the Three Effluents and Two Test Species, Lima, Ohio, 1983**

Species	Effluent	Geo. Mean
Fathead minnow	STP	>100%
	Refinery	54.8%
	Chem. plant	?
Ceriodaphnia	STP	5.5%
	Refinery	5.5%
	Chem. plant	>100%

Table 5-11. **Predicted Concentrations of STP and Refinery Effluent at Near-Field Stations Based on Conductivity Measurements, Lima, Ohio, 1983**

Station	STP	Refinery
3	66%	0%
4	44%	34%

Figure 5-4. *Ceriodaphnia* young production for ambient stations, Lima, Ohio, 1983.



STP and below the refinery as well as downstream to Station 8 as the effluent data would suggest. Thus from both sets of tests the *Ceriodaphnia* data predict impact in the river below the STP and downstream until Station 8 where the toxicity disappears.

The fathead minnow data for the chemical plant effluent cannot be easily interpreted. Since there was no clear dose-response curve, one cannot attribute the toxicity observed, which was not severe, to the chemical plant effluent.

The fathead minnow ambient toxicity data could be explained by a slightly toxic substance in the water that entered above Station 2. Indeed, in the 1982 study, a toxic slug was observed to have entered above Station 2. Since the 1983 study was performed on one grab sample, a clear-cut explanation cannot be readily provided.

6. *Dilution Analysis of the Sewage Treatment Plant, Refinery, and Chemical Plant, 1982*

A dye study was performed at each of three sites (Table 6-1) to identify the individual dilution characteristics of each effluent. By modeling downstream dilution contours for each discharger, the exposure concentration pertinent to instream effects could then be quantified.

6.1 Sewage Treatment Plant (STP)

The STP is located on the right (northwest) bank of the Ottawa River at approximately RK 60.5 and has a nominal flow of 0.53 m^3/sec (12 mgd). During the six days onsite, 20-25 September, daily average flows varied from 0.335 to 0.417 m^3/sec (Table 6-2). On 20 and 21 September, during the STP dye study, daily average flows were 0.358 and 0.36 m^3/sec, respectively. Plant-operational data during 22-25 September, provided by the STP, indicate that flows usually decrease by several mgd during the early morning hours, reaching a minimum between 0600 and 0800 hours, and then increase until 1200 hours. The STP was not as consistent during the afternoon with flows either maintaining their level recorded at 1200 hours, or decreasing until 1800 hours. The STP has storage capacity sufficient to allow the discharge flow to be regulated independently of the inflow under a variety of conditions.

Injection of Rhodamine WT dye started at approximately 1400 hours on 20 September and continued until 1545 hours on 21 September. The two Fluid Metering Inc. precision metering pumps were connected to a 200-gm/liter container of dye and a 400-gm/liter solution of $Na_2S_2O_3$, respectively. The line from the dye was inserted through the side wall of the larger line from the $Na_2S_2O_3$ and both lines were lowered down a manhole approximately 36 m from the STP discharge into the Ottawa River. The resulting dye injection rate was calculated to be approximately 5.0 ml/min. The $Na_2S_2O_3$ injection rate of 240 ml/min is equivalent to a 4.45 ppm concentration in a discharge flow of 0.36 m^3/sec, which would protect the dye from a chlorine residual of 0.7 ppm. During the two-day study, the chlorine residual was nominally held below 0.2 ppm.

The instream water samples were collected on 21 September from 1045 to 1440 hours at the 13 transects described in Table 6-1. The observed background fluorescence was 0.05 ppb at the upstream velocity transect (152 m above the discharge) and 0.3 ppb in the STP discharge prior to dye injection. The background fluorescence applied to the transect data was extrapolated between these two values in proportion to the observed dye concentration in each sample. The dye injection rate and average plant flows for 20 and 21 September result in calculated average discharge dye concentrations of 46.5 and 46.1 ppb, respectively.

Table 6-1. Transect Locations for the Dye Dilution Analysis at Three Sites on the Ottawa River, 1982 Survey

Transect	Distance from STP (m)	Transect	Distance from Refinery (m)	Transect	Distance from Chemical Plant (m)
T0a	-107				
T0b	-40				
T1	0				
T2	15				
T3	30				
T4	76				
T5	137				
T6	213				
T7	305				
T8	457				
T9	762				
		T0	-30		
	853	T1	0		
		T2	15		
		T3	30		
T10	930	T4	76		
		T5	137		
		T6	213		
		T7	305		
				T0a	-61
				T0b	-30
	1,265		410	T1	0
				T2	9
				T3	30
		T8	457		46
				T4	76
				T5	137
				T6	213
T11	1,524		671		259
		T9	762	T7	351
		T10	1,067	T8	655
		T11	1,524	T9	1,113
				T10	1,311
				T11	1,433

The average dye concentration measured at the point of discharge was 48.8 ppb on 20 September from 1430 to 2400 hours and 53.3 ppb on 21 September from 0000 to 1530 hours. The predicted concentrations are in reasonable agreement with measured concentrations considering that they were based on daily average plant flows. During the early morning hours the reduced plant flow (0.20 m³/sec at 0633 on 21 September) results in greater discharge dye concentrations. On 21 September it is likely that the actual flow for the period 0000-1530 hours was sufficiently below the daily average flow to account for the observed 15 percent increase in dye concentration. The average dye concentration recorded at the discharge between 1130 and 1230 hours on 21 September, the time when Transects T1-T4 were sampled, was 49.6 ppb. This value was used to form all dilution ratios since it accurately represents conditions while the near-field transects were being sampled and is a compromise between higher early morning values and lower daily average values for downstream transects which have a longer time history. The resulting dilution contours are shown for the surface (Figure 6-1) and for the bottom (Figure 6-2).

For depths less than 0.5 m, the same value (0.2 m from bottom) is used in each figure. Multiple depths were recorded primarily at Transects T0a and T0b upstream of the discharge and at T1 at the discharge. These figures indicate that the freshwater inflow at the top of the pool is overridden by a surface layer of

Figure 6-1. Surface dilution contours in the Ottawa River downstream from the STP, 21 September 1982.

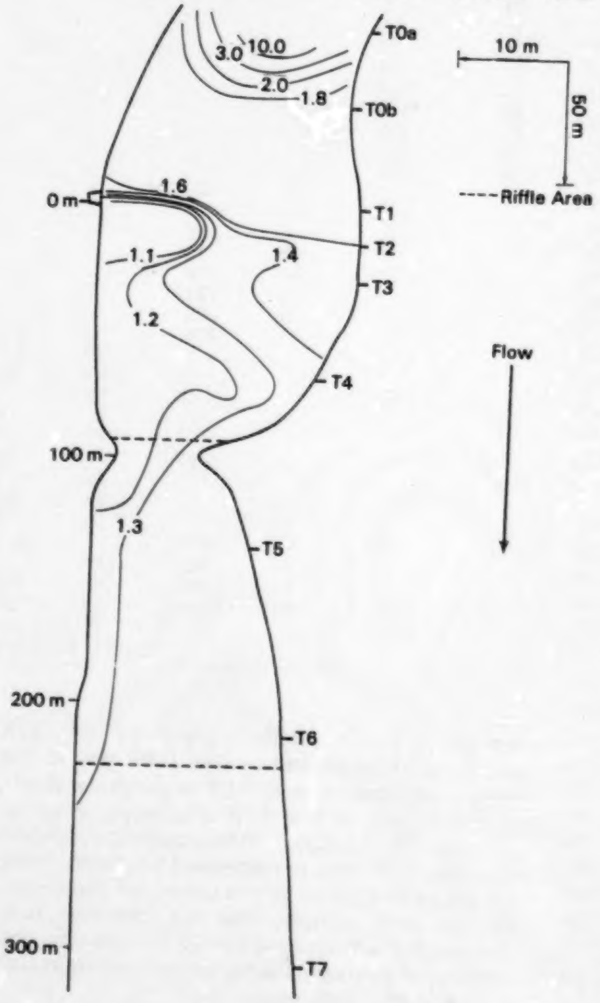
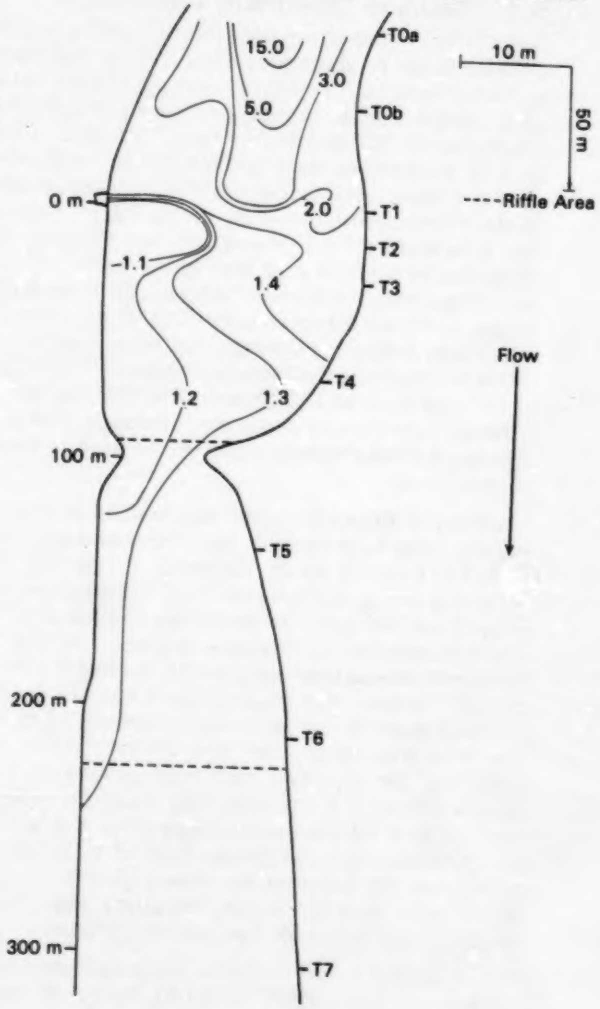


Figure 6-2. Bottom dilution contours in the Ottawa River downstream from the STP, 21 September 1982.



STP effluent, which extends upstream of the discharge. On the surface, a dilution contour of 2 extends 45 m upstream toward the head of the pool, whereas on the bottom a dilution factor of 2 extends downstream to the discharge. The effluent remained at a dilution less than or equal to 1.2 (83 percent of river flow) along the right bank for 120 m. After the first riffle area, approximately 90 m downstream of the discharge, the effluent mixed across, approaching the fully-mixed state at a dilution ratio of 1.3 (77 percent of the river flow about 210-245 m downstream). On 21 September, a flow of 0.071 m³/sec was measured upstream of the plant. This flow value, coupled with the daily average plant flow of 0.361 m³/sec, implies that the STP makes up 83.6 percent of the downstream flow during these seasonal flow conditions of the Ottawa River (Table 6-3). It is believed that the STP flow was less than the daily average value during and preceding the survey, indicating that the smaller value of 77 percent calculated from the dye study reflects these conditions.

6.2 Refinery

The refinery discharge is located on the left (southeast) bank of the Ottawa River 850 m below the discharge from the STP (approximately RK 59.7). The refinery has a nominal flow of 0.24 m³/sec (5.5 mgd). The discharge flow is recorded on a 7-day circular chart from which the values every three hours were tabulated for the period 20-25 September. The daily average flow during this 6-day period ranged from 0.201 m³/sec on Tuesday to 0.099 m³/sec on Saturday (Table 6-2). During any given day, the flow was fairly steady except on Thursday, 23 September, when the flow decreased after 1200 hours.

The injection of Rhodamine WT dye started at approximately 1210 hours on 22 September and continued to 1410 hours on 23 September. The metering pump was connected to a 70.7 gm/liter container of dye, injected just above the weir in the discharge canal at a location approximately 210 m from the river. Dye was injected at a rate of 4.96 ml/min. Midway between the weir and the river was a small basin where the continuous discharge fluorometer was installed just ahead of the outlet culvert.

Table 6-2. **River Flows Upstream of the STP and Reported Discharge Flows at Each Site, 1982 Survey**

Date	Flow (m³/sec)			
	Upstream	STP	Refinery	Chemical Plant
20 Sept.		0.358	0.187	0.068
21 Sept.	0.071	0.361 (N.A.)[a]	0.199	0.055
22 Sept.		0.335	0.201	0.059
23 Sept.	0.123	0.372 (0.218)[a]	0.173 (0.189)[a]	0.064
24 Sept.		0.417	0.106	0.062
25 Sept.	0.137	0.358 (0.277)[a]	0.099	0.069

[a] Average flow for period preceding and during initial instream sampling when different from daily average.

Note: N.A. = not available.

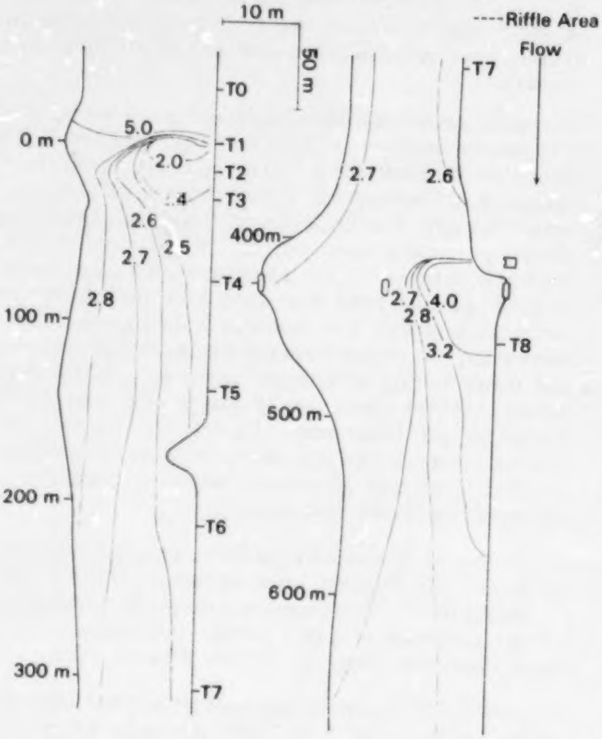
Instream water samples were collected along the transects described in Table 6-1 from 0930 to 1230 hours on 23 September. Average background fluorescence at Transect T0, upstream of the discharge, was 0.36 ppb. The background fluorescence in the discharge canal prior to initiation of dye injection was 0.15 ppb. A flow-weighted background fluorescence of 0.29 ppb was used downstream of the discharge for most samples. The average discharge dye concentration calculated from the dye injection rate and the three hourly discharge flows was 30.35 ppb between 1500 hours on 22 September and 1200 hours on 23 September. During the same time interval, the average dye concentration recorded at the discharge was 30.10 ppb, which is in excellent agreement with the calculations.

Due to the uniform discharge flow, the discharge dye concentration showed little variability during and preceding the instream survey. An average discharge concentration of 30.4 ppb (0000-1030 hours on 23 September) was used to form the dilution ratios.

Dilution contours downstream from the discharge are presented in Figure 6-3. These contours indicate a prominent discharge jet extending at right angles to the bank. The discharge influence extends to the opposite bank at the point of discharge with uniform surface and bottom values. The high initial mixing may be aided by a fallen tree, which lies across the river 23 m below the discharge. An eddy was observed, with water movement along the obstruction toward the discharge bank then upstream to be re-entrained in the discharge. The remaining mixing occurred more slowly with the discharge approaching fully-mixed at a dilution ratio of 2.65 approximately 365 m below the point of discharge. This ratio corresponds to the effluent comprising 38 percent of total flow. The effluent was diluted by the chemical plant discharge at 410 m and the river again became fully-mixed at 760 m.

To compare the observed dilution ratio with the flows, adjustments need to be made to the daily average values in order to represent survey conditions. The discharge flow decreased noticeably after 1200 hours, so an average flow of 0.189 m³/sec from 0000 to 1200 hours was used. The discharge flow of the STP from 0515 to 1030 hours was too low to be recorded properly on the 15-minute computer printout. The daily minimum flow of 0.16 m³/sec was reported at 1019 hours and all flows during this period (0515-1030 hours) were most likely less than

Figure 6-3. Dilution contours in the Ottawa River downstream from the refinery, 23 September 1982.



0.22 m³/sec, the value used here. These flows, coupled with a measured stream flow of 0.123 m³/sec above the STP, combine to give a flow of 0.531 m³/sec below the discharge during the instream survey. The flow (0.189 m³/sec) is 35.6 percent of this total flow, illustrating that the dye results are in reasonable agreement with the observed flows (Table 6-3). At Transect T11, 1,524 m below the discharge, a gradient exists with dilution increasing from mid-channel to the banks. This lack of equilibrium is probably an artifact of the reduction in flow occurring in the early morning hours at the STP. The higher dilution ratios observed near the bank are from water parcels which were passing the discharge before the early morning flow reduction occurred at the STP.

6.3 Chemical Plant

The chemical plant discharge is located on the left (southeast) bank of the Ottawa River, approximately RK 59.2. The discharge is 410 m below the refinery discharge and 1,265 m below the STP. The chemical plant discharge has a nominal flow of 0.088 m³/sec (2 mgd). From 20 to 25 September, the reported daily average flow ranged from 0.055 to 0.069 m³/sec (Table 6-2). The flow at the chemical plant discharge is nominally steady over a day.

The injection of Rhodamine WT dye started at 1045 hours on 24 September and continued to 1440 hours

Table 6-3. Ottawa River Flow and Percent Flow Contribution from the Discharges on the Days of the Three Dye Surveys, 1982 Survey

	Flow (m³/sec)	Percent Flow Contribution[a]			
		Upstream	STP	Refinery	Chemical Plant
21 Sept. (STP Survey)					
Below STP	0.432	16.4	83.6		
Below refinery	0.632	11.2	57.2	31.6	
Below chemical plant	0.686	10.3	52.7	29.0	8.0
23 Sept. (Refinery Survey)					
Below STP	0.495 (0.342)[b]	24.9 (36.2)	75.1 (63.8)		
Below refinery	0.668 (0.531)	18.5 (23.3)	55.6 (41.1)	25.9 (35.6)	
Below chemical plant	0.732 (0.594)	16.9 (20.8)	50.8 (36.7)	23.6 (31.8)	8.7 (10.7)
25 Sept. (Chemical Plant Survey)					
Below STP	0.495 (0.414)	27.6 (33.0)	72.4 (67.0)		
Below refinery	0.594 (0.513)	23.0 (26.7)	60.3 (54.0)	16.7 (19.3)	
Below chemical plant	0.662 (0.582)	20.7 (23.5)	54.0 (47.7)	15.0 (17.0)	10.3 (11.8)

[a] Values also represent percent contribution of effluent at fully-mixed zone for each discharger.
Example: Below STP on 21 September:
83.6 percent flow contribution x 49.6 ppb STP discharge concentration
= 41.5 ppb fully mixed concentration 738 m downstream.

[b] Average flow for period preceding and during initial instream sampling when different from daily average.

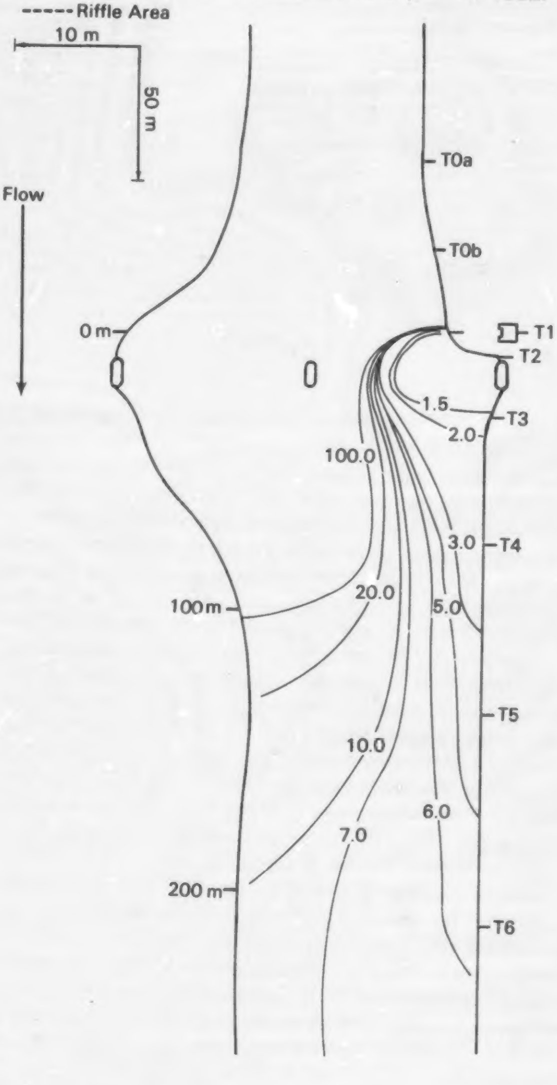
on 25 September. At approximately 1245 hours on 24 September the metering pump was reset to give a lower injection rate. The metering pump was connected to a 70.7 gm/liter container of dye, injected into a culvert across the road from the discharge house and approximately 60 m from the river. The continuous discharge fluorometer was installed at the discharge house. Monitoring of dye weight showed a dye injection rate of 2.86 ml/min. Instream water samples were collected on 25 September between 0915 and 1240 hours along the transects described in Table 3-1. Average background fluorescence at Transects T0a and T0b was 0.19 ppb, whereas the background fluorescence in the chemical plant discharge prior to the dye injection was 0.05 ppb. A flow-weighted background fluorescence of 0.17 ppb was used downstream from the discharge. The daily average discharge flow on 24 and 25 September of 0.062 and 0.069 m^3/sec results in a predicted dye concentration at the point of discharge of 54.6 and 49.0 ppb, respectively. The recorded discharge dye concentration was 53.5 ppb on 24 September (1300-2330 hours) and 48.9 ppb on 25 September (0000-1430 hours). An average discharge dye concentration of 49.9 ppb (0000-1030 hours), representative of the preceding time history, was used in forming dilution ratios. The dilution ratios downstream of the chemical plant discharge are presented in Figure 6-4.

The plume remained along the left bank for approximately 120 m before beginning to mix the rest of the way across. From 460 to 760 m, the river is approximately fully-mixed at a dilution ratio of 7.1, although the far bank (right) continues to display slightly higher dilution (i.e., lower dye concentrations). A dilution ratio of 7.1 corresponds to the chemical plant effluent that is 14 percent of the river flow. The total river flow below the chemical plant discharge corresponding to its time history is 0.582 m^3/sec. This number is composed of 0.137 m^3/sec measured above the STP, 0.277 m^3/sec at the STP during the morning hours, and the daily average flows at the refinery and chemical plant of 0.099 and 0.069 m^3/sec, respectively. These reported flow values correspond to the chemical plant effluent making up 11.8 percent of the river flow on 25 September (Table 6-3).

6.4 Evaluation of Dilution Characteristics

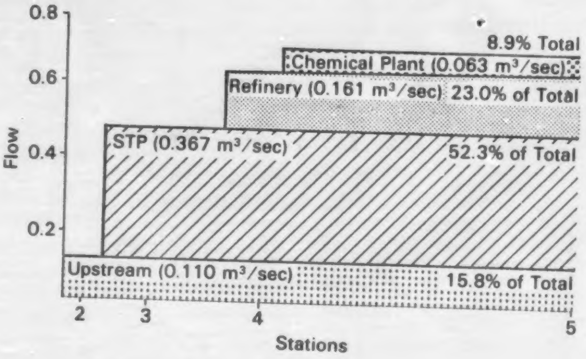
The Ottawa River flow and the fully-mixed (percent) flow contribution below each discharge from each upstream source is summarized in Table 6-3 for the dates of the three dye surveys. Table 6-3 also reflects percent concentration of effluent at the fully-mixed zone for each of the dischargers. Individual contributions vary among days, primarily as a result of the upstream flow increasing from 0.071 to 0.137 m^3/sec, whereas the refinery discharge decreased from approximately 0.20 to 0.10 m^3/sec during the study period. As discussed in Sections 5.1-5.3, the contribution at any given time is different from the daily average value due to the hourly flow variations at the STP. The effluent contributions to the receiving water based upon the average flows observed during the study period, 20-25 September, are illustrated in Figure 6-5.

Figure 6-4. Dilution contours in the Ottawa River downstream from the chemical plant, 25 September 1982.



Under the low upstream flow condition of the study, the STP strongly influenced the pool into which it discharged. Above the discharge more dilution occurred on the bottom because of the upstream inflow,

Figure 6-5. Effluent contribution to receiving water.



whereas on the surface, water with a dilution of 2 extended 45 m towards the head of the pool. The STP effluent was fully-mixed 245 m downstream of the discharge. The average STP flow composed 83 percent of the river on the study day and decreased to 72 percent four days later as a result of increased upstream inflow. The refinery discharge had a prominent jet at right angles to the bank with the majority of the mixing occurring in the first 60 m. The river gradually became fully-mixed 365 m below the discharge, just above the chemical plant discharge. During the days of the three dye studies, the refinery discharge ranged from 16.7 to 31.6 percent of the river flow, whereas the STP ranged from 55.6 to 60.3 percent of the river flow below the refinery. For the first 120 m downstream from the chemical plant discharge, this effluent was confined to one-third of the river's width. The majority of the mixing of the chemical plant effluent with the STP/refinery water occurred by 245 m, and the river gradually approached the final fully-mixed state 760 m downstream from the discharge. Here the chemical plant effluent comprised 8-10 percent of the river flow, whereas the refinery contributed 15-29 percent and the STP 51-54 percent during the days of the three dye studies.

Additional flow measurements in the study area are available for four days in September 1979 (Engineering-Science 1981). These data indicate that the average upstream river flow above the STP flow was 0.487 m³/sec, whereas the average STP flow was 0.459 m³/sec. Under these conditions of increased upstream flow, the fully-mixed STP effluent was 49 percent of the river flow. During the same period, the average flow measured at Allentown, Ohio was 1.44 m³/sec. The United States Geological Survey reports a 44-year average discharge at Allentown of 3.57 m³/sec (USGS 1976). The USGS data for water-year 1975 indicates monthly average December - March flows ranging from 5.04 to 15.46 m³/sec, whereas flows in summer and fall ranged from 0.75 to 2.05 m³/sec. These data indicate that the flows reported by Engineering-Science are probably representative of typical fall conditions, whereas the low upstream flows of 0.071 to 0.137 m³/sec observed by EPA more likely represent a low-flow event.

A low-flow 7Q/10 event cannot be addressed directly since a 7Q/10 flow is not available for the Ottawa River above the STP. Under any worst case condition, the STP flow will dominate the river flow such that at any point below the STP discharge the upstream river flow could be approximated by the STP flow (i.e., assume zero upstream flow). During the 6-day study period, average flows at the STP, the refinery, and the chemical plant were 0.367, 0.161, and 0.063 m³/sec, respectively. Under these conditions, the STP effluent would make up 100 percent of the river above the refinery (93 percent with a 0.028 m³/sec (1 cfs) upstream flow). Downstream of the refinery, the river would be 70 percent STP and 30 percent refinery. Downstream of the chemical plant, the river would be 62 percent STP, 27 percent refinery, and 11 percent chemical plant. Only the STP contribution would change by more than 1 percent if a 0.028 m³/sec (1 cfs) upstream flow had been used.

7. *Periphytic Community, 1982 Survey*

An effect on the periphytic community may be seen in either a reduction of an important habitat or food source for invertebrates and fish or the enhancement or dominance of nuisance species of algae that neither support lower trophic levels nor are aesthetically pleasing. The following discussion is intended to present an overview of the response of the periphytic community to the discharges. Support periphyton data on the composition and abundance are presented in Appendix Tables D-1 and D-2. Methods used in the periphyton studies are in Appendix Section C.1.

7.1 Community Structure

Periphyton communities at each station were numerically dominated either by diatoms or green algae (Table 7-1). Station 1 at Thayer Road was characterized by low periphyton density, high diversity, and high equitability relative to the other sampling locations (Figure 7-1). Total density increased nearly three-fold at Station 2 upstream from the Lima Sewage Treatment Plant (STP), diversity was the highest observed at any station (partly because of the greater number of genera observed), and equitability declined somewhat although it was still moderately high. Composition of dominant taxa was similar to that observed at Station 1.

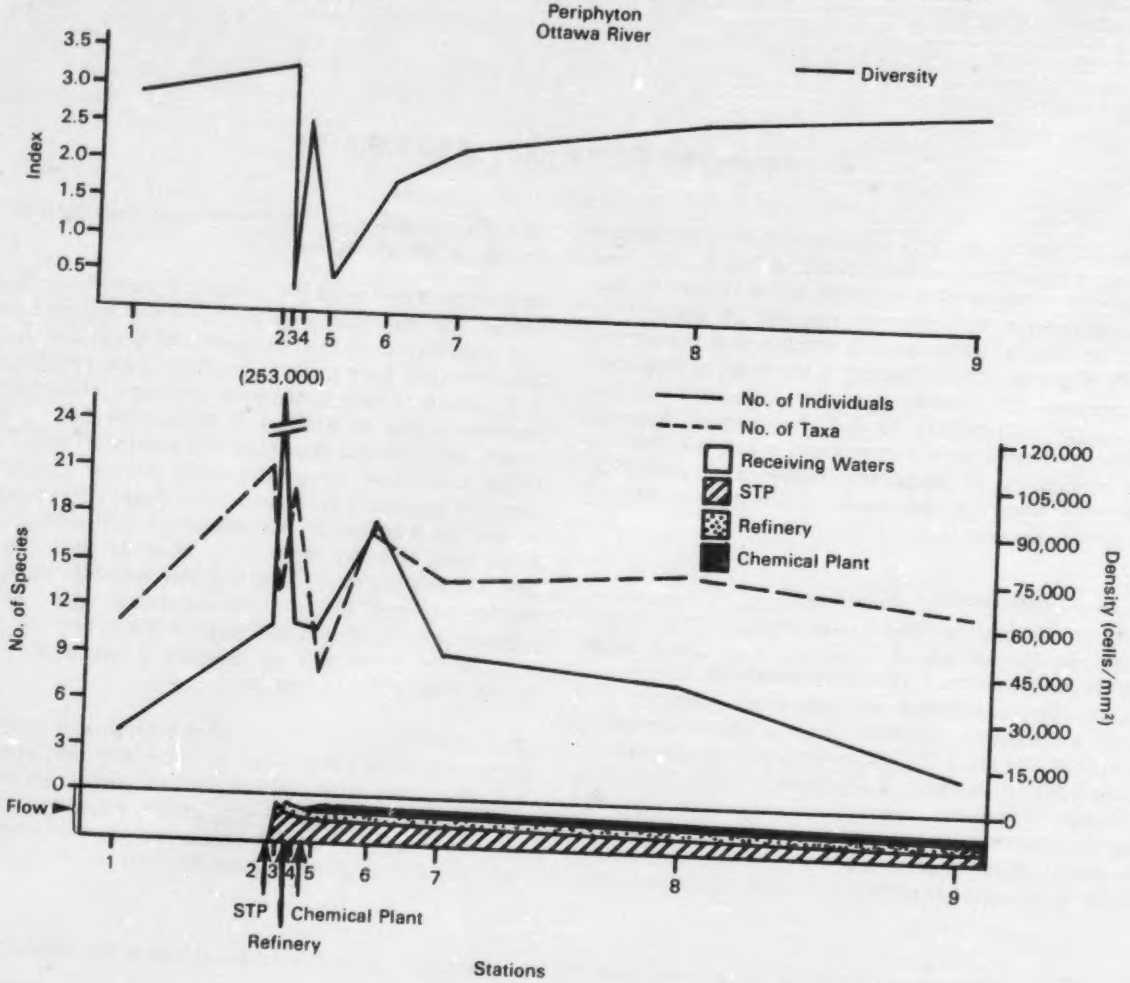
At Station 3 between the Lima STP and the refinery, periphyton abundance exhibited a pronounced five-fold increase over that observed at Station 2. All of this increase was caused by *Stigeoclonium* (Figure 7-2), and as a result diversity and equitability were extremely low at Station 3. Except for blue-green algae, which were absent at this station (Table 7-1), other taxa that were important at the upstream stations persisted at reduced densities. Total density at Station 4 between the refinery and the chemical plant was very similar to that observed at Station 2 (56,000 cells/mm²). Diversity and equitability were moderate, and the diatom *Nitzschia* was clearly numerically dominant at Station 4. Several other taxa that were important at Station 2 were at least moderately abundant at Station 4.

Immediately downstream of the chemical plant at Station 5, total periphyton density was unchanged, but the green alga *Stigeoclonium* was once again abundant. Diversity and equitability were again very low, although values for these indices were not as low as those observed at Station 3. Periphyton

Table 7-1. **Summary of Periphyton Composition, Diversity, and Standing Crop on Natural Substrates in the Ottawa River, September 1982**

Parameter	Sampling Stations 1	2	3	4	5	6	7	8	9
Density (cells/mm²)									
Diatoms	12,670	33,022	7,382	42,295	2,494	25,137	30,824	32,321	7,382
Green algae	3,791	13,367	245,488	9,079	52,570	62,544	10,475	6,285	4,290
Blue-green algae	2,095	8,679	0	4,289	0	2,195	4,988	1,796	898
Total Periphyton	18,556	55,168	252,970	55,763	55,064	89,876	46,287	40,402	12,570
Percentage of Total Density									
Diatoms	68.28	59.86	2.92	75.85	4.53	27.97	66.59	80.00	58.73
Green algae	20.43	24.33	97.04	16.28	95.47	69.59	22.63	15.56	34.13
Blue-green algae	11.29	15.73	0.00	7.69	0.00	2.44	10.78	4.45	7.14
Taxa (Genus) Diversity ($\bar{d}$)	2.91	3.29	0.28	2.63	0.42	1.76	2.25	2.71	2.92
Taxa (Genus) Equitability (e)	0.95	0.67	0.09	0.45	0.19	0.25	0.46	0.60	0.81
Total Genera Identified	11	21	13	19	8	17	14	15	13
Chlorophyll *a* (mg/m²)	31.9	273.6	296.5	193.8	111.7	151.0	166.6	135.9	102.4
Biomass (g/m²)	22.2	61.8	35.9	29.8	19.5	39.6	19.6	23.2	18.7
Autotrophic Index (Weber 1973)	971	230	135	176	216	269	119	225	208

Figure 7-1. Spatial distribution of periphyton community indices and associated parameters, 1982 survey.



composition was considered similar at Stations 3 and 5 because *Stigeoclonium* dominated, diatoms composed less than 5 percent of total density, and blue-green algae were absent at these sampling locations (Table 7-1). At Station 6 further downstream from the chemical plant, total density increased to approximately 90,000 cells/mm². Diversity and equitability also were higher than observed at Station 5, but values for these indices of community structure were still moderately low. *Stigeoclonium* numerically dominated the periphyton at Station 6, but the diatoms *Achnanthes*, *Navicula*, and *Nitzschia* were also important components of the community.

At Station 7 near Allentown, total density declined by approximately 50 percent, whereas diversity and equitability increased to moderate levels. *Stigeoclonium* declined in abundance but remained an important component of periphyton. The community was dominated by *Nitzschia*, although *Navicula* and filamentous blue-green algae were also abundant (Table D-1). Twenty kilometers farther downstream at Station 8, total density remained similar to that observed at Station 7, but diversity and equitablity increased to moderately high values. *Navicula* was the most abundant genus; *Nitzschia* and *Stigeoclonium* were also important periphyton constituents (Table D-1).

At Station 9, the farthest downstream station, periphyton abundance was similar (<20,000 cells/mm²) to that recorded at the upstream control (Station 1). Diversity at these stations was essentially identical (2.92 vs. 2.91), and equitability at Station 9 was high (Table 7-1). There were, however, differences in composition between Stations 1 and 9, although *Navicula* and *Nitzschia* were important at both stations (Table D-1).

7.2 Chlorophyll *a* and Biomass

Chlorophyll *a* standing crop in the Ottawa River ranged from approximately 32 to 296 mg/m²; biomass standing crop varied from 19 to 62 g/m² (Table 7-1). Chlorophyll *a* was significantly lower at Station 1 than at all other stations. The high Autotrophic Index (AI) value at Station 1 combined with the low chlorophyll *a* values indicated periphyton either was primarily composed of heterotrophic (nonalgal) taxa or contained a relatively large proportion of nonliving organic matter (APHA 1981). However, low algal standing crop (as indicated by chlorophyll *a*) was more responsible for this high AI value than any increase in heterotrophic or nonliving biomass. Based on data from the other sampling locations, AI values less than approximately 250 appeared to be typical for the Ottawa River in this September survey.

Maximum biomass standing crop during this survey occurred at Station 2 where chlorophyll *a* standing crop was near maximum (Table 7-1). At Station 3 below the Lima STP, chlorophyll *a* reached maximum value, but biomass standing crop exhibited a 40 percent decline from that observed at Station 2. Both chlorophyll *a* and biomass progressively declined at Stations 4 and 5 and then exhibited increases at Station 6. Chlorophyll *a* continued to increase at Station 7, whereas biomass decreased to values near those observed at Station 1 (reference station). Chlorophyll *a* progressively declined at Stations 8 and 9, but biomass standing crop remained at approximately 20 g/m².

7.3 Evaluation of Periphytic Community Response

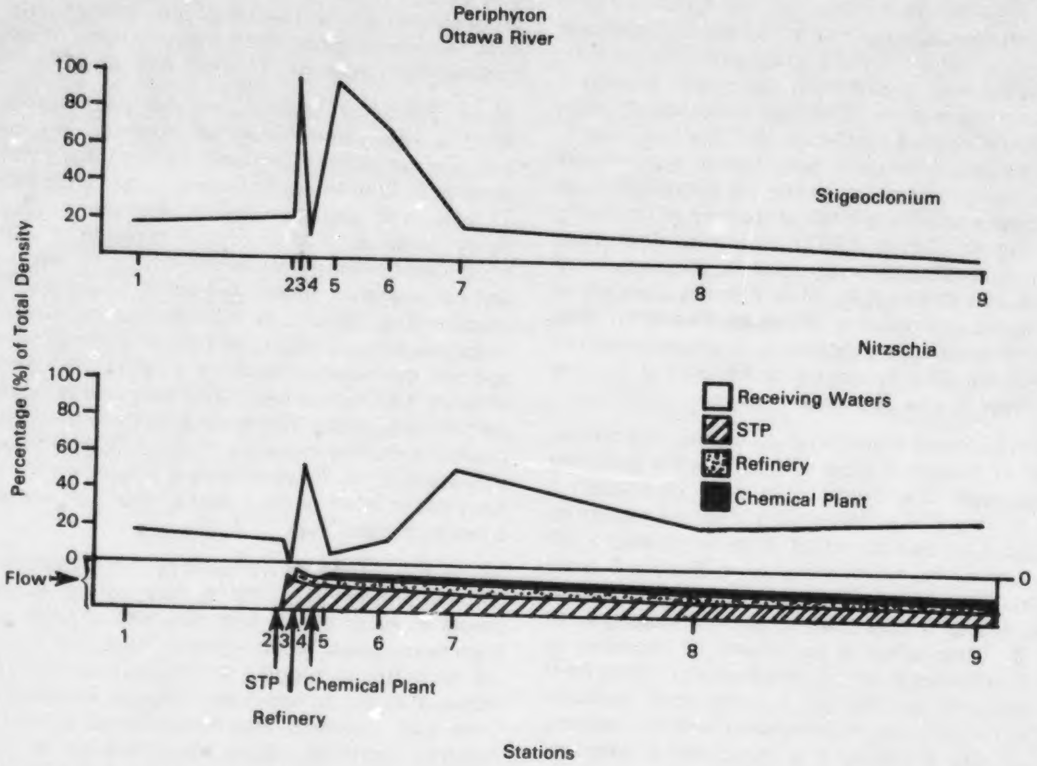
The apparent responses of the periphyton community to the three principal discharges into the Ottawa River varied depending on the parameter(s) being evaluated. For chlorophyll *a* and biomass data, any responses (either separate or additive) to the three discharges need to be evaluated with an understanding of the general health of Station 2, located immediately upstream from the Lima STP. Station 2 as a reference station exhibits effects from some upstream source associated with the City of Lima. The greatest biomass and diversity were found at this station along with a relatively high chlorophyll *a* value. *Skeletonema, Thalassiosira,* and *Amphora* were present in greatest abundance at Station 2. Predominance of certain species of these genera suggests effects other than nutrient enrichment, such as organic loading. Adverse effects of discharges on periphyton often cause large increases in the Autotrophic Index (Weber 1973; APHA 1981). However, nutrient enrichment caused by discharges from waste treatment facilities also are considered effects altering natural populations. The discharge from the Lima STP caused a reduction in the AI index, suggesting that periphytic algae were responding positively to nutrients contained in the discharge. Subsequent dilution and depletion of these nutrients was reflected in progressively increasing AI values at downstream stations (Stations 4, 5, and 6).

More distinct responses to the discharges were evident when periphyton abundance, composition, and diversity were examined. Except for the very high density at Station 3 and the moderately high density at Station 6, total periphyton abundance was relatively uniform at Stations 2 through 8 (40,000-55,000 cells/mm²). However, diversity, equitability, and periphyton composition fluctuated greatly. *Stigeoclonium* and *Nitzschia* exhibited the most discernable response to the discharges; both of these genera have species considered tolerant of polluted conditions (Palmer 1977); however, they respond differently to certain pollutants. The discharge from the Lima STP caused a drastic increase in the density and relative abundance of *Stigeoclonium*, apparently at the expense of other genera, and diversity plummeted as a result (Figure 7-2).

Below the discharge from the refinery, *Nitzschia* and *Stigeoclonium* exhibited a reversal of abundance levels to each other, and diversity approached the high level observed upstream from the Lima STP. These patterns suggest a mitigation by the refinery effluent to the nutrient enrichment imposed by the Lima STP. However, this mitigation of effects by the refinery discharge upon the influence of the STP effluent is considered temporary and localized because results at Station 5 reflect a return to affected conditions prior to the refinery discharge. The persistence of effluent constituents (i.e., inorganic nutrients) from the STP is considered the primary factor influencing the periphyton rather than a similar effect from the effluent of the chemical plant because (1) the contribution of the chemical plant's discharge to the river flow is inconsequential compared to that of the Lima STP (Table 6-3), and (2) the chemical plant's discharge was primarily non-contact cooling water during the course of the study. Recovery occurred in a progressive fashion as illustrated in Figure 7-2. *Nitzschia* did not exceed *Stigeoclonium* in relative abundance until Station 7, which was approximately 13 km downstream from the chemical plant and approximately 5.6 km downstream from the Shawnee STP. The periphyton community showed continued recovery at Station 8, and density, composition, and diversity at Station 9 were as similar to those at Station 1 as could be expected even for an unaffected stream over such a long distance (72 km).

In summary, periphyton communities in the Ottawa River were numerically dominated by either diatoms or green algae. The green alga *Stigeoclonium* and the diatom, *Nitzschia*, exhibited the greatest response to the three major discharges. Data on periphyton

Figure 7-2. Spatial distribution of key periphyton taxa, 1982 survey.



abundance, composition, and diversity were more useful for evaluating the effects of the discharges than the chlorophyll *a* and biomass data, although these latter data were useful in determining that the major effect of the Lima STP was probably nutrient stimulation rather than organic loading or toxicity. Effluent from the Lima STP produced the greatest periphyton response, as was expected from the large volume of this discharge. Some mitigative improvement of conditions is provided temporarily from the refinery discharge. Uncertainty remains concerning the effects of discharge from the chemical plant. No major additive effects of the three principal discharges were observed. Periphyton at downstream stations showed substantial recovery toward conditions existing upstream from the Lima STP. Comparison of the periphytic communities at Stations 7 and 8 did not indicate nutrient loading from the Shawnee STP located between the stations to have a similar effect as that exhibited by the Lima STP.

8. Benthic Macroinvertebrate Community, 1982 Survey

The benthic community is considered a good indicator of ambient response to adverse conditions because of their general lack of extensive mobility. The degree of community stability within affected areas can be measured by comparing composition and dominance to that of nonaffected areas. An effect on the benthos would be apparent as an alteration in community structure, standing crop, or species composition of the benthos beyond the limits of normal fluctuation within the receiving waterbody. The increased abundance of nuisance insect larvae or other benthic species also would be regarded as an effect. The following discussion is intended to present an overview of the response of the benthic community and selected populations to the discharges. Support benthic data on the composition, relative abundance, and community parameters are presented in Appendix Tables D-3 and D-4. Methods used for benthos are discussed in Appendix Section C.2.

8.1 Community Structure

Composition and abundance of benthic invertebrates varied between stations as summarized in Table 8-1 (based on the 38 most abundant taxa). The community at Station 1, the upstream reference station (above the City of Lima) was dominated by the trichopterans *Cheumatopsyche* and *Hydropsyche*, which together comprised about 43 percent of individuals. The variety of taxa found at this station were relatively high; however, total faunal abundance was the lowest of all stations sampled. Reference Station 2, which was downstream of the City of Lima but above the three discharges, had many taxa in common with Station 1, but had a substantially different community based on dominance. At Station 2, the chironomid *Cricotopus bicinctus* grp. was most abundant, representing 31 percent of the individuals in the community.

A major shift in community dominance was observed at Station 3, below the Lima Sewage Treatment Plant (STP) discharge. At this location simuliids (blackflies) were overwhelmingly dominant, with the larvae (both unidentified Simuliidae and *Simulium*) representing almost 63 percent of the fauna, and the pupae comprising another 11 percent of the individuals collected (Table 8-1). The total family thereby accounted for almost three quarters of the community.

The community at Station 4, below the refinery discharge, was again remarkably different from Station 3. Community composition shifted back to dominance by the chironomid *C. bicinctus* grp., although to a greater extent than seen at Station 2. At Station 4, this taxon comprised 55 percent of the individuals collected. However, in addition to compositional shifts, total density of fauna at this station also dropped dramatically compared to Station 3, and was similar to that seen at Station 1. Relative composition of taxa at Station 5, below the chemical plant discharge was similar to that at Station 4, except that the dominance of *C. bicinctus* grp. was stronger (representing 70 percent of the individuals collected), and total abundance of individuals was much greater. The benthic community composition and dominance at Station 6 was similar to that at Station 5, although the absolute density of the dominant chironomid had decreased by about one third. At Station 7, which was located below the Shawnee STP, dominance of the benthic community shifted back to the simuliids as had been observed at Station 3. The chironomid *C. bicinctus* grp. was also relatively abundant at this station. Although Station 7 had many taxa in common with Station 3, individuals were more evenly distributed among a greater number of taxa at Station 7. At Stations 8 and 9 community composition tended to be closer to that observed at Station 1, with dominance by the trichopterans *Cheumatopsyche* and *Hydropsyche* and an increased abundance of a greater variety of taxa including the ephemeropterans. However, at Station 8, the chironomid *C. tremulus* was the second most abundant taxa. Also, total faunal abundance was much greater at Stations 8 and 9 than at Station 1. At Station 9, the abundance of hydropsychids increased so much that the abundance and distribution of other taxa decreased compared to Station 8.

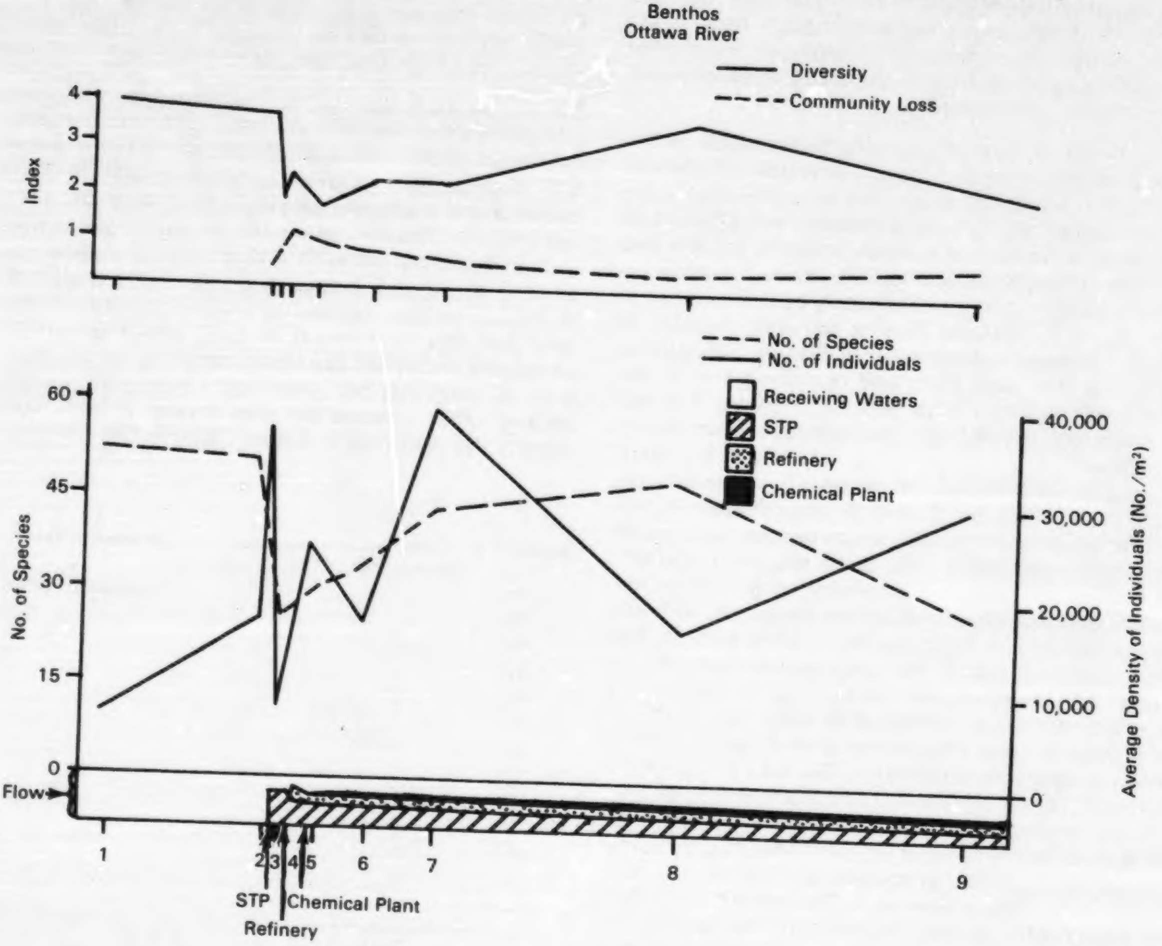
Community response was summarized by examining an index of diversity based on information theory and a community loss index based on composition. Diversity and number of species and number of individuals are graphed by station in Figure 8-1. Diversity of the benthic community shows a response to location of the discharges within the study area. Diversity was greatest (3.895 calculated on log base 2) at reference Station 1, above the City of Lima, and decreased only slightly at reference Station 2, below

Table 8-1. **Average Density (No./m²) of the Most Abundant Species at Each Sampling Station, Ottawa River, 21 September 1982**

Station	STA1		STA2		STA3		STA4		STA5		STA6		STA7		STA8		STA9			
Species	Number Indivs	PCT Comp	Number Indivs	PCT Comp	Number Indivs	PCT Comp	Number Indivs	PCT Comp	Number Indivs	PCT Comp	Number Indivs	PCT Comp	Number Indivs	PCT Comp	Number Indivs	PCT Comp	Number Indivs	PCT Comp	Number Total	PCT Comp
Simuliidae, L	15.82	0.22	1220.40	7.14	24082.56	62.05	49.72	0.66	18.08	0.07	135.60	0.80	25605.80	64.66	22.60	0.14	90.40	0.30	5359.62	24.59
C. (Cricotopus) Bicinctu	18.08	0.25	5299.70	31.00	1572.96	4.05	4178.74	55.44	17736.48	70.08	11048.58	65.10	4271.40	10.79	696.08	4.25	18.08	0.06	4857.69	22.28
Cheumatopsyche, L	1717.60	24.10	854.28	5.00	388.72	1.00	0.00	0.00	0.00	0.00	0.00	0.00	1469.00	3.71	4370.84	26.68	12434.52	41.52	2435.02	11.17
Hydropsyche, L.	131.08	1.84	0.00	0.00	0.00	0.00	0.00	0.00	9.04	0.04	0.00	0.00	90.40	0.23	1814.78	11.08	7957.46	26.57	1161.01	5.33
Thienemannimyia, Grp	210.18	2.95	940.16	5.50	2178.64	5.61	1152.60	15.29	3249.88	12.84	511.33	3.01	858.80	2.17	133.34	0.81	0.00	0.00	1041.96	4.78
Hydropsychidae, L	1360.52	19.09	65.54	0.38	54.24	0.14	2.26	0.03	0.00	0.00	11.30	0.07	881.40	2.23	2063.38	12.59	3597.92	12.01	913.72	4.19
Stenelmis L.	339.00	4.76	1789.92	10.47	1109.66	2.86	33.90	0.45	36.16	0.14	0.00	0.00	768.40	1.94	668.96	4.08	2024.96	6.76	769.45	3.53
Simuliidae, P	2.26	0.03	45.20	0.26	4246.54	10.94	0.00	0.00	0.00	0.00	0.00	0.00	158.20	0.40	0.00	0.00	0.00	0.00	514.02	2.36
C. (Cricotopus) Tremulus	20.34	0.29	90.40	0.53	612.46	1.58	266.68	3.54	0.00	0.00	22.60	0.13	293.80	0.74	2660.02	16.23	18.08	0.06	455.94	2.09
Chironomidae, P.	126.56	1.78	277.98	1.63	142.38	0.37	784.22	10.40	1780.88	7.04	226.00	1.33	226.00	0.57	169.50	1.03	0.00	0.00	423.62	1.94
Elmidae, L.	174.02	2.44	1211.36	7.09	768.40	1.98	18.08	0.24	18.08	0.07	0.00	0.00	226.00	0.57	237.30	1.45	687.04	2.29	383.15	1.76
Baetis, N.	770.66	10.82	334.48	1.96	0.00	0.00	0.00	0.00	36.16	0.14	11.30	0.07	610.20	1.54	180.80	1.10	922.08	3.08	318.77	1.46
Empididae, L.	13.56	0.19	137.86	0.81	2183.16	5.63	4.52	0.06	99.44	0.39	0.00	0.00	90.40	0.23	99.44	0.61	54.24	0.18	309.83	1.42
Polypedilum (S.S.) Convi	45.20	0.63	0.00	0.00	36.16	0.09	117.52	1.56	280.24	1.11	406.80	2.40	768.40	1.94	300.58	1.83	180.80	0.60	221.01	1.01
Nanocladius, L.	9.04	0.13	1184.24	6.93	45.20	0.12	81.36	1.08	65.54	0.26	135.60	0.80	67.80	0.17	183.06	1.12	18.08	0.06	203.40	0.93
Caenis, N.	124.30	1.74	1066.72	6.24	47.46	0.12	0.00	0.00	0.00	0.00	0.00	0.00	22.60	0.06	363.86	2.22	0.00	0.00	188.42	0.86
Stenelmis, A.	31.64	0.44	58.76	0.34	72.32	0.19	0.00	0.00	81.36	0.32	22.60	0.13	67.80	0.17	513.02	3.13	614.72	2.05	167.92	0.77
Polypedilum (S.S.) Scala	11.30	0.16	162.72	0.95	47.46	0.12	0.00	0.00	0.00	0.00	375.73	2.21	971.80	2.45	18.08	0.11	0.00	0.00	153.21	0.70
Bothrioneurum Vejdovskya	248.60	3.49	228.26	1.34	192.10	0.49	268.94	3.57	117.52	0.46	0.00	0.00	0.00	0.00	2.26	0.01	216.96	0.72	148.21	0.68
Ephemeroptera, N.	393.24	5.52	94.92	0.56	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	293.80	0.74	153.68	0.94	379.68	1.27	146.11	0.67
IMM Tubif with Cap Chaet	0.00	0.00	0.00	0.00	2.26	0.01	81.36	1.08	262.16	1.04	915.30	5.39	180.80	0.46	4.52	0.03	0.00	0.00	142.70	0.65
IMM Tubif w/o Cap Chaet	70.06	0.98	253.12	1.48	0.00	0.00	61.02	0.81	171.76	0.68	248.60	1.46	22.60	0.06	36.16	0.22	235.04	0.78	121.41	0.56
Diptera P.	20.34	0.29	58.76	0.34	22.60	0.06	162.72	2.16	461.04	1.82	180.80	1.07	22.60	0.06	38.42	0.23	0.00	0.00	107.74	0.49
Berosus L	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	117.52	0.46	711.90	4.19	203.40	0.51	0.00	0.00	0.00	0.00	98.81	0.45
Tricladida	58.76	0.82	786.48	4.60	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	98.28	0.45
Chironomus, L.	0.00	0.00	133.34	0.78	2.26	0.01	9.04	0.12	18.08	0.07	652.58	3.84	0.00	0.00	18.08	0.11	0.00	0.00	81.73	0.37
Physella	4.52	0.06	0.00	0.00	0.00	0.00	0.00	0.00	207.92	0.82	203.40	1.20	293.80	0.74	18.08	0.11	0.00	0.00	73.06	0.34
Baetidae, N.	122.04	1.71	56.50	0.33	0.00	0.00	0.00	0.00	9.04	0.04	0.00	0.00	248.60	0.63	63.28	0.39	90.40	0.30	62.81	0.29
Cricotopus Syvlestris GR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	137.86	0.54	480.25	2.83	0.00	0.00	0.00	0.00	0.00	0.00	60.70	0.28
Heptageniidae, N.	27.12	0.38	9.04	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	350.30	2.14	36.16	0.12	49.14	0.23
Rheotanytarsus L.	2.26	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	241.82	1.48	144.64	0.48	45.20	0.21
Limnodrilus Udekemianus	0.00	0.00	0.00	0.00	72.32	0.19	126.56	1.68	81.36	0.32	22.60	0.13	0.00	0.00	9.04	0.06	54.24	0.18	42.05	0.19
Stenacron, N.	94.92	1.33	0.00	0.00	2.26	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	241.82	1.48	0.00	0.00	39.42	0.18
Glyptotendipes L.	2.26	0.03	54.24	0.32	228.26	0.59	9.04	0.12	0.00	0.00	0.00	0.00	22.60	0.06	18.08	0.11	0.00	0.00	38.37	0.18
Tanytarsus L.	61.02	0.86	18.08	0.11	126.56	0.33	0.00	0.00	0.00	0.00	0.00	0.00	22.60	0.06	99.44	0.61	0.00	0.00	37.58	0.17
Hydroptila, L.	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	339.00	0.86	42.94	0.26	0.00	0.00	36.53	0.17
Simulium, L.	0.00	0.00	0.00	0.00	282.50	0.73	0.00	0.00	0.00	0.00	0.00	0.00	22.60	0.06	0.00	0.00	0.00	0.00	34.95	0.16
Potamothrix Bavaricus	0.00	0.00	20.34	0.12	0.00	0.00	6.78	0.09	72.32	0.29	180.80	1.07	45.20	0.11	0.00	0.00	0.00	0.00	32.59	0.15
Other Species	899.48	12.62	644.10	3.77	289.28	0.75	122.04	1.62	241.82	0.96	468.95	2.76	432.23	1.09	551.44	3.37	171.76	0.57	423.36	1.94
Station Total and Date Total	7125.78		17096.89		38808.71		7537.10		25309.74		16972.60		39598.05		16384.99		29947.26		21798.44	

Note: L. = Larva S.S. = *Sensu strictu* (in the strict sense)
P. = Pupa A. = Adult
N. = Nymph
Capitalization of taxa is due to computerized format.

Figure 8-1. Spatial patterns of benthic species diversity and components of diversity, Ottawa River, 1982 survey.



Stations

the City of Lima. However, diversity of the community below the Lima STP, at Station 3, dropped substantially, to 1.753. A slight improvement in diversity occurred at Station 4, below the refinery discharge, but was caused by a high evenness or relatively few individuals distributed among relatively few taxa. A return to the minimum observed diversity was then seen at Station 5, below the chemical plant discharge. Recovery in terms of community diversity progressed from Station 6 through 8 with diversity at Station 8 (3.605) close to that observed at the upstream reference stations. Diversity at Station 9, the farthest downstream recovery station, was lower than that at Station 8.

The community loss index from Courtemanch (1983) is based on the presence or absence of species and emphasizes taxonomic differences between the reference station and the station of comparison. The premise behind the index is that rarer species are given equal weight to the more abundant taxa. Therefore, an effect is measured as the elimination of entire species populations. The formula for community loss is as follows:

$$I = \frac{A-C}{B}$$

where

A = number of species found at reference station
B = number of species found at station of comparison
C = number of species common to both stations

As the value increases, the degree of dissimilarity with the reference station increases. The spatial trend in the values illustrates a peak in the index at

Station 4, although the values at Stations 3, 5, and 6 are similarly high (Figure 8-1). These data from the community loss index suggest the greatest effect upon the benthic community composition occurred at Station 4 where results of the diversity index indicated some "false" improvement.

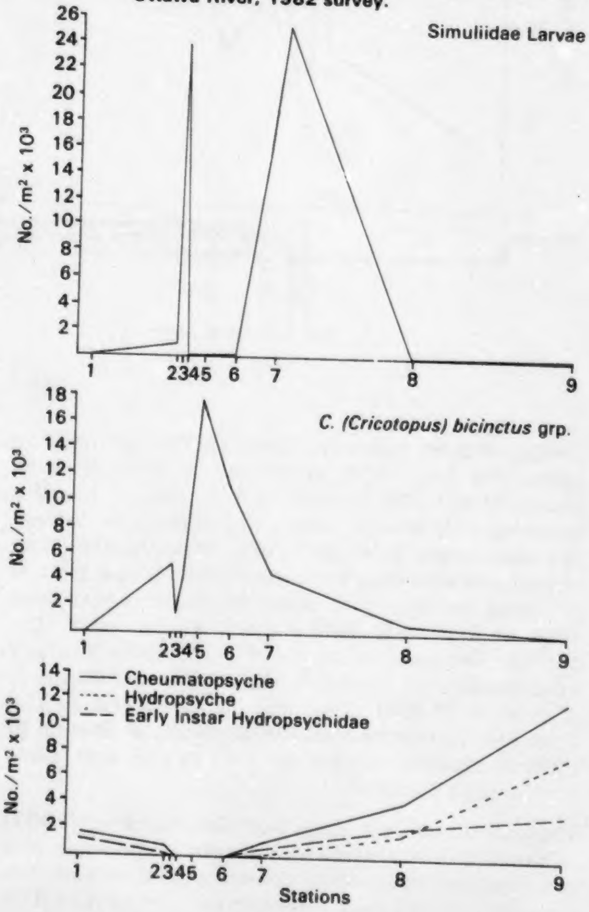
The pattern of diversity is reflected strongly in the evenness component of the diversity index (Appendix Table D-4) which considers the way individuals are distributed among species. Evenness and richness, or the relative number of species present, are the two primary components of diversity, while the community loss index is influenced solely by the number of species. In the Ottawa River study area, number of species dropped substantially at Station 3 (Figure 8-1), below the Lima STP, and remained low in the vicinity of all three dischargers (i.e., Stations 3, 4, and 5). Therefore, the effect on the number of species was consistent among the affected stations. The pattern of recovery at Station 4, below the refinery discharge, relative to Stations 3 and 5 is reflected in the evenness component and can be best understood by examining number of individuals present (Figure 8-1). At Station 3, despite the drop in number of species, total abundance of individuals increased substantially primarily due to an increase in only a few species. The high dominance of a few species corresponds (by definition) to a low evenness, and therefore low diversity. Although number of species remained low at Station 4, total abundance also dropped back to levels comparable to reference Station 1 (Figure 8-1), so that the predominance of one or a few species was not as strong, and evenness and diversity both increased. As the community loss index suggests, the composition of the community at Station 4 is dissimilar from that at Station 1. The pattern described for Station 3 (i.e., predominant abundance of one or a few species) reoccurred at Station 5, resulting in a low evenness component and low diversity. Downstream of Station 5 the patterns were more subtle. Number of species (i.e., richness component) recovered from Stations 6 through 8, as did the evenness component and diversity. Although total abundance of individuals was high at Station 7, below the Shawnee STP, predominance by one (or a few) species was apparently not as strong as at Stations 3 and 5, so that the effect on diversity was small. The combination of a drop in number of species and a slight decrease in evenness resulting from an increase in number of individuals was responsible for the decline in diversity observed at Station 9.

8.2 Spatial Trends in Key Taxa

Certain key taxa represent the greatest contribution to total abundance of the benthic community evaluated under diversity and its components. Based on the results presented in Table 8-1, Simuliidae larvae, the midge, *C. bicinctus*, and the hydropsychids, *Cheumatopsyche* and *Hydropsyche*, were the numerically dominant taxa and their abundance trends exert the major apparent effect on the spatial fluctuations in abundance of the total benthic community.

Simuliidae increase slightly in abundance at Station 2 from the upstream reference station and then reach maximum abundance at Stations 3 and 7, below each STP (Figure 8-2). At the intermediate stations between 3 and 7, abundances were low, never exceeding 140/m². The overwhelming dominance of simuliids in combination with a decrease in number of species at Station 3 caused a low diversity and is reflected in the higher redundancy value. This response was much less at Station 7 because number of species remained at a level indicative of recovery, and the remaining individuals were evenly distributed among other species so that diversity was high despite the dominance of Simuliidae at that station.

Figure 8-2. Spatial abundance patterns of key benthic taxa, Ottawa River, 1982 survey.



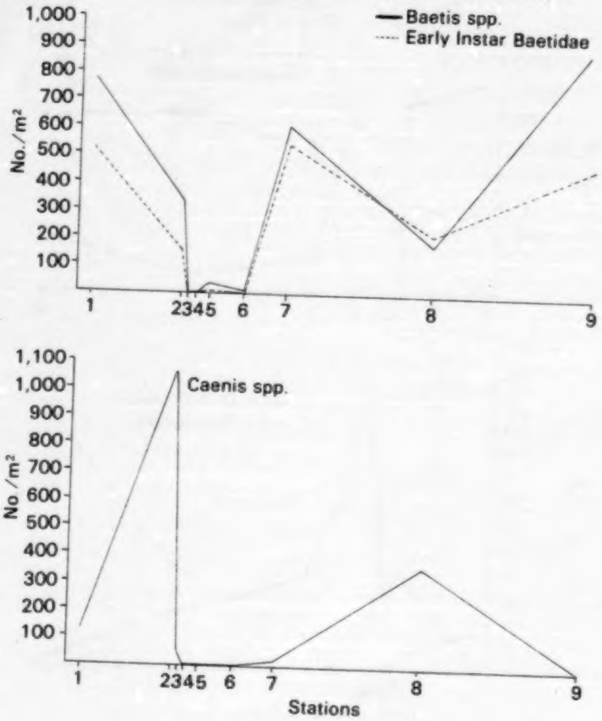
Cricotopus bicinctus grp. occurred in maximum abundance at Station 5 below the chemical plant discharge and decreased steadily at following downstream stations (Figure 8-2). The numerical dominance of *C. bicinctus* and the low number of species at Station 5 accounted for low diversity values at that station. *C. bicinctus* also showed a secondary abundance peak at Station 2, with a subsequent decline at Station 3 below the Lima STP.

The numerically dominant caddisflies *Cheumatopsyche* and *Hydropsyche* exhibited similar abundance trends to each other; their spatial distribution indicated adverse water quality conditions between Lima STP and Allentown. They dropped in abundance after the Lima STP and did not increase in numbers until Station 7 (Figure 8-2). Recovery continued downstream where peak densities of both genera were found at Station 9.

Although the Ephemeroptera constituted a smaller component of the benthic community than either the dipterans or trichopterans, the spatial trends in abundance of the mayflies showed a strong relationship to the location of the discharges, thus depicting similar trends as other groups. Both *Caenis* and *Baetis* along with early instar Baetidae were nearly absent from Station 3 below the Lima STP discharge, through Station 6, downstream of all three dischargers (Figure 8-3). Recovery began at Station 7. Interestingly, *Baetis* and *Caenis* showed converse abundance patterns at stations other than the discharge stations, which may indicate competitive interaction between the two genera.

Spatial trends of major benthic groups are plotted for comparison with those of the key taxa to evaluate the relative quantity and quality of information gained regarding instream effluent effects. Ephemeroptera and Trichoptera were present in widely different densities and are plotted on different abundance scales; however, they underwent similar abundance trends relative to location of the plant effluents, being nearly depleted from the stations nearfield to the dischargers (Figure 8-4). The spatial fluctuations of both of these groups are reflective of the key taxa within those groups. Therefore, comparable inferences of community response would be drawn from examination of these two major taxonomic groups as from examination of component species. The Chironomidae and Oligochaeta, also graphed on different density scales from each other both depict a somewhat different response to the dischargers than that of the ephemeropterans and trichopterans (Figure 8-4). Both the midges and worms decrease in abundance immediately after the Lima STP (Station 3), and then begin to increase in abundance, with Chironomidae peaking at Station 5 and Oligochaeta at Station 6. Both groups then undergo a decrease in abundance downstream with the exception of a rise in abundance for the Oligochaeta at Station 9. This pattern suggests an opportunistic response (i.e., increase in abundance) to stressed conditions beginning below the City of Lima (Station 2) and continuing more strongly in the vicinity of the discharges, with this pattern interrupted by an apparent toxic response after the Lima STP (Station 3).

Figure 8-3. Spatial abundance patterns of the dominant ephemeropterans, Ottawa River, 1982 survey.



The spatial trend in total Chironomidae matches the spatial trend of *C. bicinctus*, so that interpretation based on the total group would be similar to that for the dominant component species. However, the spatial trend in abundance of Oligochaeta is due to several species influencing total abundance. *Bothrioneurum vejdovskyanum* is a primary contributor to the peak densities of worms at Stations 1, 2, 4, and 9. The maximum peak of worms found at Station 6 is due to a normally uncommon species, *Potamothrix bavaricus* (including immature forms) which was not abundant at the reference and furthest downstream station (Table 8-1). Species of *Limnodrilus* also contribute to peaks at Stations 4, 5, and 9, but are not abundant at the intermediate stations. In this case, component species respond variously, not only to location of the discharges, but also to related and interactive factors such as competition, predation, food availability, and microhabitat characteristics.

Figure 8-4. Spatial distribution of major benthic groups, 1982 survey.

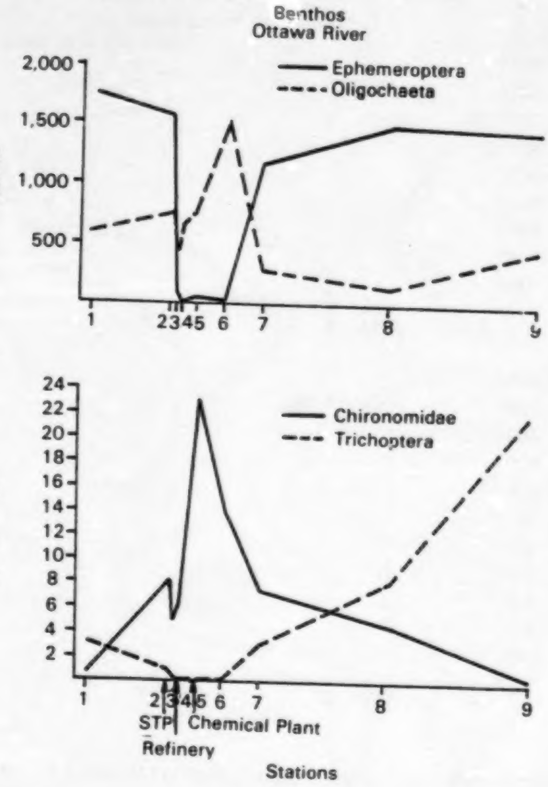


Table 8-2. Density (No./100 m³) of Macroinvertebrates Collected from the Drift, Ottawa River, 23 September 1982

Macroinvertebrate	Station 2	3	4	5	Average
Hydrozoa		1.24	0.56		0.45
Tricladida	3.01	29.07			8.02
Gastropoda				8.96	2.24
Ancylidae				5.52	1.38
Physella		0.34	0.62	23.34	6.08
Tubificidae	11.22			3.17	3.60
Naididae	16.44	9.61	72.97	5.51	26.13
Branchiobdellidae	53.27		0.30		13.39
Acarina				2.22	0.56
Cladocera	1.61	7.37	0.61		2.40
Ostracoda		0.38			0.10
Cyclopoida	30.26	5.66	128.02	450.64	153.64
Calanoida		1.34			0.34
Isopoda				0.52	0.13
Astacidae	1.40		0.86		0.56
Orconectes			0.30	1.06	0.34
Unid. Insect.		1.03			0.26
Collembola		0.52	0.30	1.59	0.60
Ephemeroptera N.		0.38		5.82	1.55
Caenis N.	8.12	2.31	1.49	1.59	3.38
Baetis N.	1.40			12.45	3.46
Heptageniidae N.	3.70				0.92
Zygoptera N.			0.62		0.16
Coenagrionidae N.			1.22	1.70	0.73
Argia N.				1.59	0.40
Calopteryx N.				0.52	0.13
Corixidae A.			0.30	0.52	0.20
Gerridae, imm.				0.52	0.13
Hydropsychidae L.	3.70	0.77			1.12
Coleoptera L.	2.80		0.56		0.84
Elmidae L.	5.11	0.77			1.47
Elmidae A.				4.32	1.08
Stenelmis L.			0.30		0.08
Stenelmis A.			1.85	5.31	1.79
Dubiraphia L.		0.38			0.10
Haliplidae A.				1.18	0.30
Dytiscidae A.				0.52	0.13
Diptera L.			1.85	0.52	0.59
Diptera P.	73.70	95.05	484.28	3,851.36	1,126.10
Simuliidae L.	19.62	34.79	5.42		14.96
Simuliidae P.		0.38			0.10
Empididae L.		1.28	0.56	1.70	0.88
Empididae P.		0.38			0.10
Chironomidae L.	143.22	95.12	411.76	1,019.11	417.30
Psychodidae L.			0.62	0.52	0.28

8.3 Benthic and Zooplankton Drift Collections

Density estimates averaged over replicates, for each of the four drift collection stations, is presented in Table 8-2. These taxa represent five major taxonomic groups: aquatic insects, zooplankters, gastropods, annelids, and (non-zooplanktonic) arthropods. Several minor taxa (e.g., Turbellaria and Hydrozoa) were also represented. Of the taxa encountered, the aquatic insects were the most abundant, with the chironomid larvae and dipteran pupae (most of which are chironomid pupae) being numerically dominant. Chironomid larvae and pupae, together, constituted approximately 86 percent of the total number of individuals collected.

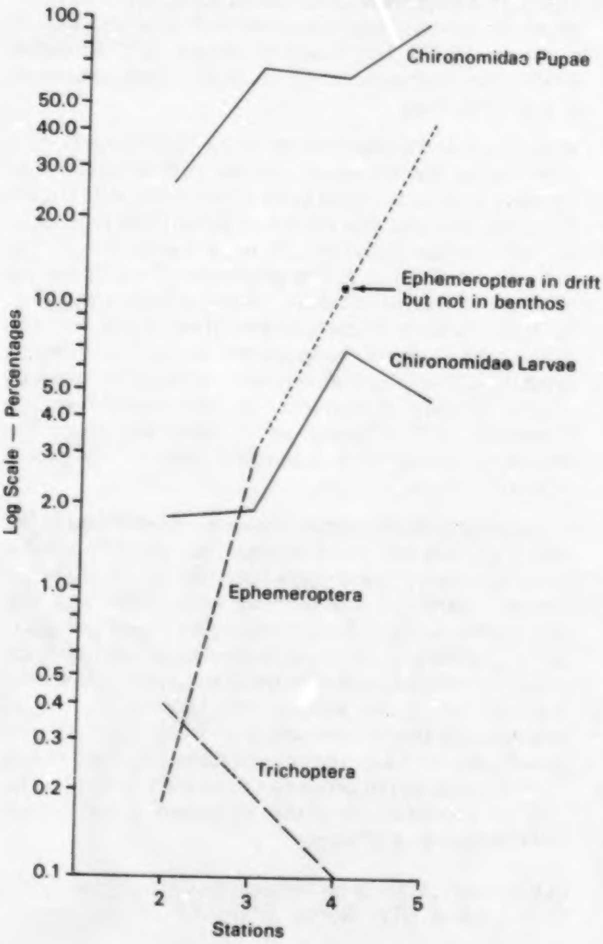
Density comparisons across stations indicate that the lowest total abundance of organisms occurred at Stations 2 and 3 (Table 8-2). Density increased at Station 4 and peaked at Station 5, with an average of 5,414/100 m³. The greater densities at Stations 4 and 5 were attributable in part to the greater abundances of chironomid larvae and pupae at these stations compared to the reference Station (2) and the STP influenced Station (3). Community diversity, in terms of the number of species present, was greatest at Station 5 where 29 taxa were identified and lowest at Station 2 where only 17 taxa were collected. The additional taxa at Station 5 (and also Station 4) were insects, primarily dipterans, odonates, and hemipterans, which were lacking at Stations 2 and 3.

The densities of the predominant taxa in the drift from each station reflected the dominance of the benthic populations at those stations which indicates that drift as a dispersal mechanism is heavily dependent upon localized drifting. The greatest density of simuliid larvae in the drift occurred at Station 3 where the largest population in the benthos was found; simuliids were completely absent in the drift from Station 5 where, again, they were rare in the benthos. Approximately 82 percent of the dipteran pupae and 78 percent of the chironomid larvae encountered were from Station 5, where they were more abundant in the benthos than at the other stations sampled for drift.

Although ephemeropterans were relatively minor components of the drift in terms of abundance contribution, individuals of this insect order were present at all four stations indicating that colonization potential of this group is not entirely eliminated from the affected area where the benthic populations of this group are not abundant. None of the insects other than Chironomidae were abundant in the drift at any station. However, more insect orders were represented at Stations 3, 4, and 5 compared to the reference Station (2). The converse was true for the benthic population; that is, more insect orders were present at Station 2 compared to the downstream stations.

The relative population contribution of major benthic taxa was compared among stations to ascertain whether a relationship between standing crop and drift abundance would show informative spatial trends. Differences of relative proportions within groups among stations might suggest adverse environmental pressures resulting in population instability. An index of proportioning population abundances in the drift was obtained simply by calculating the ratio of drift density to benthic density. A plot of these indices (percent) illustrates the spatial trend of the chironomid larvae and pupae, ephemeropterans, and trichopterans (Figure 8-5). The standing crop-drift relation indices indicated that drifting of chironomid larvae and pupae and ephemeropterans increased downstream through the affected area. Sampling constraints, such as varying time of collection at each station for drifting organisms which exhibit distinct diel periodicity, may have had some influence on spatial differences. However, increasingly adverse water quality conditions from upstream to downstream could also influence higher drift proportions. Drifting of the trichopterans steadily decreased in direct proportion to the community from upstream to downstream. Neither drifting trichopterans nor those in the benthos were found at Station 5.

Figure 8-5. Spatial trends of proportion (percentage) of population in drift compared to benthic standing crop for major taxonomic groups, 1982 survey.



Very little information on zooplankton drift was gained from this study primarily because of the large mesh (500 μm) of the nets used in the survey. However, cyclopoid copepods were an abundant component of the drift and exhibited a decrease in density from Station 2 (30/100 m^3) to minimal abundance at Station 3 (6/100 m^3), and then increased to relatively large abundances of 128/100 m^3 at Station 4 and 450/100 m^3 at Station 5 (Table 8-2). Other zooplankton components were not abundant in the drift collections with 500-μm mesh nets.

8.4 Evaluation of the Macroinvertebrate Community

The sampling design for the 1982 survey was taken from previous surveys (Martin et al. 1979; Engineering-Science 1981) so that direct comparisons of spatial community trends could be related among the three studies. The findings of the present study supported degradation of the benthic community from the Lima STP to the Allentown Dam, where initial recovery of the benthic community was noted. These findings are similar to those found in the Ohio EPA study (Martin et al. 1979). The health of the community improved downstream of Allentown. However, station-specific intricacies of the composition and diversity of the benthic community need to be evaluated using knowledge of specific organism sensitivity and ecology to better understand the effects imposed by the discharges.

A major shift in the benthic community structure occurred below the Lima STP discharge from that at the upstream reference station as reflected in an abrupt drop in diversity and number of species and a substantial increase in abundance of Simuliidae (blackflies). The reduction in number of species at Station 3 was primarily due to an absence of many insects other than dipterans and coleopterans. The trichopterans and ephemeropterans, which constitute the more important insects within this group, were noticeably lacking from Station 3. Although species within these two insect orders exhibit a wide variation in tolerance to adverse water quality conditions (Harris and Lawrence 1978; Hubbard and Peters 1978), each species is generally restricted to a finite range of water quality conditions. The distinct absence of the caddisflies and mayflies from below the Lima STP suggests a relatively high level of intolerance to the effluent constituents of the STP. Simuliid larvae are filter feeders and feed upon nutrients and planktonic organisms flowing past their places of attachment (Davies et al. 1962; Stone 1964). Some species of *Simulium* are very tolerant of organic pollution (Hilsenhoff 1981). The overwhelming dominance of Simuliidae at Station 3 suggests an area of nutrient enrichment which is limiting to other less facultative organisms.

A slight rise in the diversity index at Station 4 located between the refinery and chemical plant discharges reflected a compositional shift in the community. A high evenness value supported by a decrease in the number of species and benthic abundance at Station 4 from that observed at Station 3 influenced the diversity index (Figure 8-1). The abrupt decrease of simuliids from peak density at Station 3 to minimum abundance at Station 4 supports some alteration of the effects of the Lima STP by the refinery discharge. The community loss index suggested that the composition of the community was dissimilar to that of Station 1. This alteration of the community at Station 4 does not reflect recovery from effects of the Lima STP. The results from the drift collections indicated that drifting is relatively localized and successful colonization from an upstream source population is not occurring. Insufficient data exist to asess the effects upon the survival or propagation of those populations experiencing effluent plume entrainment during drifting period.

Engineering-Science (1981) found inconsistent spatial abundance trends in their zooplankton drift data among seasons, even though zooplankton abundances were higher below the Lima STP than above in September of 1979 which corresponds to the month of our collection. No attempt was made by Engineering-Science to separate out the component taxa so a direct comparison to our data is not warranted. Results of zooplankton data from the present study's drift collections suggest that the cyclopoid copepods, which are the numerically dominant macrozooplankton component, are not able to maximize population potential until after the refinery discharge.

The benthic community exhibited stressed conditions at Stations 5 and 6 located 3.4 and 8.0 km downstream, respectively, from the Lima STP. However, the community dominants were different at these two stations. *Cricotopus bicinctus*, a midge larva, dominated the fauna at Station 5, while *Potamothrix bavaricus*, a tubificid worm, dominated at Station 6. This difference in benthic structure may have been more influenced by subtle habitat differences than by variations in water quality. The habitat at Station 6 was characterized by extensive algal mats attached to the substrate. Also, more sediment was sampled at Station 6 allowing for a more complete sampling of infauna than at Station 5 where a rockier substrate was present. Most species of *Cricotopus* are considered saproxenous (tolerant of slightly polluted waters) (Beck 1977) and their presence is not surprising; however, the specific reason for their dominance at Station 5 is not clear. *P. bavaricus*, although a relatively uncommon species of worm, is found in various river systems in the United States (Spencer 1978). Not much is known of their water quality requirements, and their dominance at Station 6 is unexplained.

A recovery of the community at the Allentown Dam is exemplified by the return of the trichopterans and ephemeropterans along with other insects. It should be noted that the abundance of Simuliidae increased at this station (Station 7) to a level above that collected at Station 3. The proximity of the Shawnee STP upstream from Station 7 is most likely the reason for this increase. In comparison of effects, both STPs (Lima and Shawnee) apparently contribute nutrient enrichment to the receiving waters; however, toxicity to the benthic community is not present in the Shawnee STP effluent as is apparent from the relative stability of the benthic community at the Allentown Dam.

Comparison of the spatial trends in abundance of the major groups and the key taxa indicate that similar results and conclusions are obtained when the major groups examined are dominated by relatively few taxa. Information is lost in relying on major groups in terms of diversity indices and associated components, which are based on the number of taxa and distribution of individuals among the taxa. It should be emphasized that examining trends of major benthic groups should be at the lowest possible level (family taxonomic level) in order to retain as much information on composition shifts as possible for correct interpretation of effects.

In summary, effects on the benthos are primarily due to the Lima STP. Some alteration of the benthic community occurs below the refinery discharge, but not enough for recovery of the community. Colonization potential from drifting is low in the area below the Lima STP. Recovery of the benthic community was determined to occur at Allentown located approximately 14 km downstream from the Lima STP. Some nutrient enrichment may occur at Allentown due to the Shawnee STP, but no ambient community response to toxicity was determined.

9. *Fish Community, 1982 Survey*

The fish community is the highest trophic level to be potentially affected by polluted discharges. It ultimately represents the major concern as a sport fishing resource and reflects the environmental health of the stream. The following discussion is intended to be an overview of the response of the fisheries community to the selected discharges. Support data to this study are included in Appendix Tables D-5 and D-6. The fisheries methods are detailed in Section C.3.

9.1 Community Structure

The fisheries collections in the Ottawa River yielded 27 species, one hybrid, and three taxa of fish that could only be identified to the family or genus level due to their small size (Table 9-1). In total, seven families were represented. The most widely distributed fish were the fathead and bluntnose minnows, creek chub, and redfin shiner, which were collected at seven of the nine stations. The green sunfish and bluegill were each caught at six stations.

At the reference station (Station 1) above the City of Lima, 12 species representing four families were caught, with redfin shiner dominating the catch. An abundance of young-of-the-year or juvenile shiners (*Notropis*) were found. Four species of darters also were found to be common.

Station 2 was located below Lima, but above the three discharges examined in this study. Fifteen species from four families were found at this station. The

Table 9-1. **Results of Fisheries Survey of Ottawa River, Abundance by Station, 24-26 September 1982**

	Station								
	1[a]	2[a]	3[a]	4	5	6	7	8	9
Gizzard shad									2
Carp		2							
Cyprinidae (small)									121
Golden shiner		20	5				1	1	
Fathead minnow		38	1	15	6		2	50	4
Creek chub	4	21	81	5			1	242	4
Spotfin shiner								29	227
Emerald shiner							1	1	32
Bluntnose minnow	130	98	10	2			4	1,249	319
Stoneroller	16	2						3	
Sand shiner								3	8
Redfin shiner	1,098	3,227	3,430		2		3	26	42
Notropis sp.	444							3	6
White sucker			36	1					
Black redhorse			16						
Golden redhorse	2								
White catfish		1							
Tadpole madtom									1
Blackstripe topminnow									10
Rock bass	2								
Green sunfish		118	25	2	2			3	2
Bluegill	10	227	23	1			3		2
Largemouth bass	1	19	3						2
Black crappie		5							
Lepomis sp.							4		2
Lepomis x *Lepomis* hybrid		9							
Greenside darter	23	6							(3)[b]
Rainbow darter	22		1						
Fantail darter	16	2							
Johnny darter	28								1
Blackside darter									1
No. of species	13	15	11	6	3	0	8	11	18

[a] Aliquot procedures used.
[b] Qualitative sample separate from standard unit of effort.

darters were much more poorly represented both in numbers and diversity than at Station 1, whereas minnows, particularly the redfin shiner and sunfish predominated.

The general abundance and number of species captured remained high at Station 3, located just below the Lima Sewage Treatment Plant (STP) discharge; however, the presence of darters was reduced to only one specimen of rainbow darter. Suckers, on the other hand, appeared in moderate numbers compared to the two reference stations.

General abundance and diversity were further reduced below the refinery outfall at Station 4. Only 26 fish from three families were found at this station. Three species totaling 10 fish were caught at Station 5, downstream from the chemical plant discharge. Several species of minnows and the green sunfish were the predominant fish at both stations. No fish were caught at Station 6, located approximately 6 km downstream of the refinery discharge. This station was heavily fouled with filamentous algae.

The minnow family predominated at the stations farther downstream. Sunfish reappeared but in low numbers. A total of 19 fish from seven species were caught at Station 7, indicating some recovery, despite low DO levels. Nine species of minnows totaling 1,610 fish plus three green sunfish were caught at Station 8.

At Station 9, approximately 58 km downstream from the three discharges, species variety had risen back to levels found at Stations 1 and 2 with a greater number of families represented at this furthest downstream station. Darters once again appeared here, although in very low numbers with only one Johnny darter and one blackside darter caught. Three additional greenside darters were found in a qualitative sample outside the station collection effort.

The Shannon-Wiener diversity index and the community loss index were performed on the catch data from the fisheries survey at Ottawa River. Diversity increased at Stations 4 and 5 to a level comparable to the reference station (Station 1). These results could be misleading in suggesting recovery of the fish community in this area. However, the high diversity values were due to high evenness and low redundancy, or, as illustrated by number of species and abundance (Figure 9-1), few individuals being distributed among few species. Diversity decreased at Station 3 from that observed at the reference stations. Although abundance remained high at 3, the number of species decreased resulting in this lowered diversity. The combination of these data suggests that some alteration of the fish community below the STP has occurred.

The community loss index showed a strong dissimilarity between the communities at the influenced stations of 4, 5, and 6 to that of the reference station (Figure 9-1). These results supported the conclusions derived by the trend in the diversity index in that the fish community was apparently most affected by the refinery discharge. Any direct effects from the STP on the fish community is subtle and minimal in comparison to the community changes noted below the refinery.

The salient trends in spatial distribution are illustrated by the major components of the fish community (Figure 9-2). Two species of minnows account for the largest proportion of fish abundance at the sampling stations. The redfin shiner was most abundant and numerically dominant upstream from the refinery discharge. The redfin shiner was not abundant downstream of the refinery and had not returned to former abundance levels noted at reference areas by Kalida. Conversely, the bluntnose minnow was not abundant above the refinery, but became numerically dominant at Stations 8 (Rimer) and 9 (Kalida).

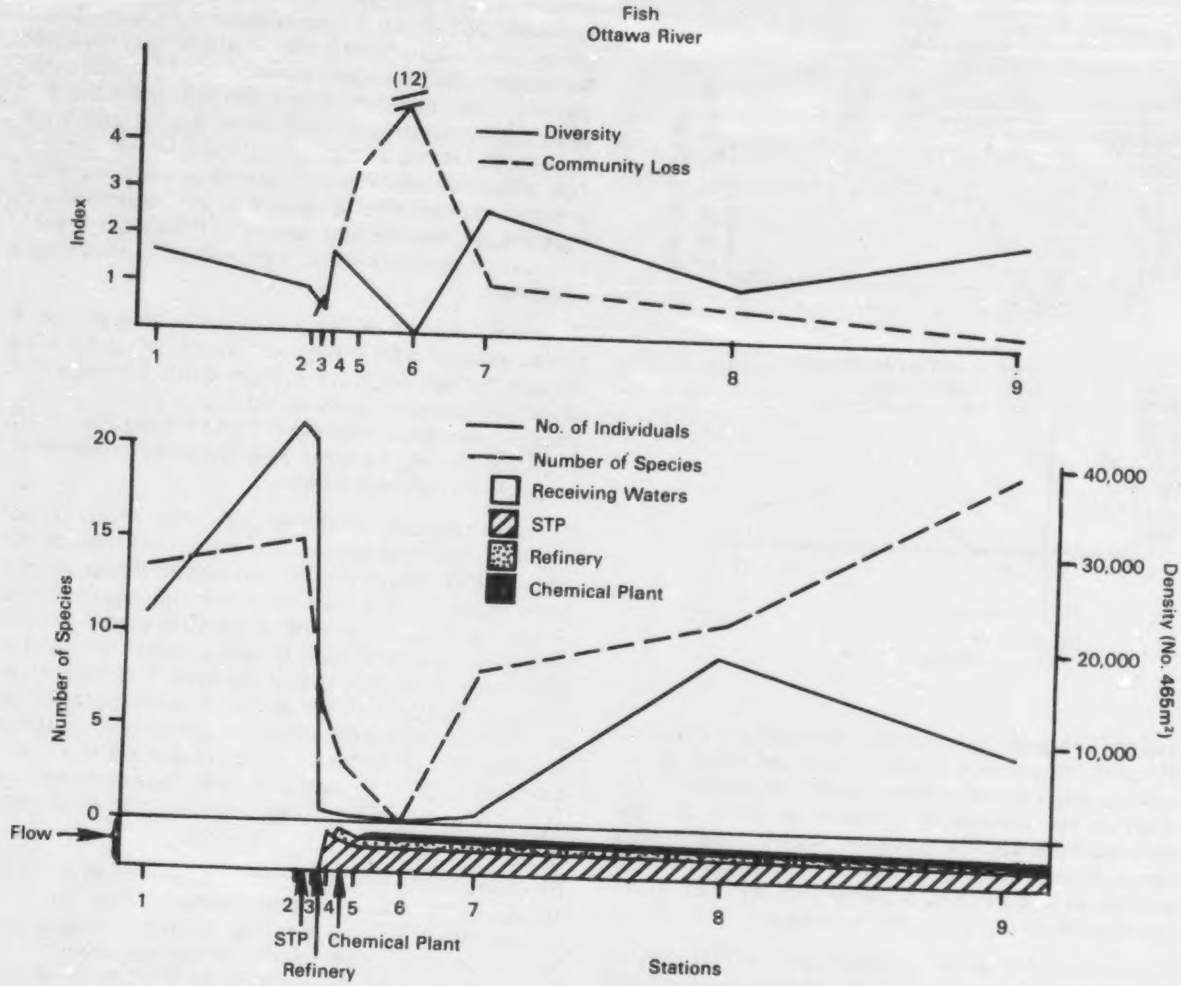
The substantial decrease in the presence of darters from the reference area above Lima to the affected area had a considerable influence on the number of species at each station (Figure 9-2). Four species of darters comprising 89 individuals were found above Lima. The darter population appeared to be affected prior to the STP and never recovered until Kalida where only three species in low abundance were collected. One of the species, the blackside darter, was found at no other station. Centrarchids also had a major influence on the spatial trend of species numbers (Table 9-1) which declined steadily from 15 species at Station 2 above the STP to zero at Station 6 (Figure 9-2) approximately 5 mi downstream from Station 2. The variety of fish was restored partially at Station 7 (below Allentown Dam) and increased to maximum levels (18 species) at Kalida, the furthest downstream station.

9.2 Evaluation of Fish Community Response

Fish collections were made on the Ottawa River during 1976 and 1977 by the Ohio Environmental Protection Agency as part of a water quality study (Martin et al. 1979). The methods used in the EPA study were qualitative in nature, but the results can be used to compare trends in the fish communities found in this study. Collections were made by Ohio EPA at Stations 1, 2, 4, 7, 8, and 9 plus several others not sampled during our study.

Due to the greater intensity of the Ohio EPA's sampling efforts, more species were found than in the present survey. Most notably, EPA encountered the grass pickerel (*Esox americanus vermiculatus*) at

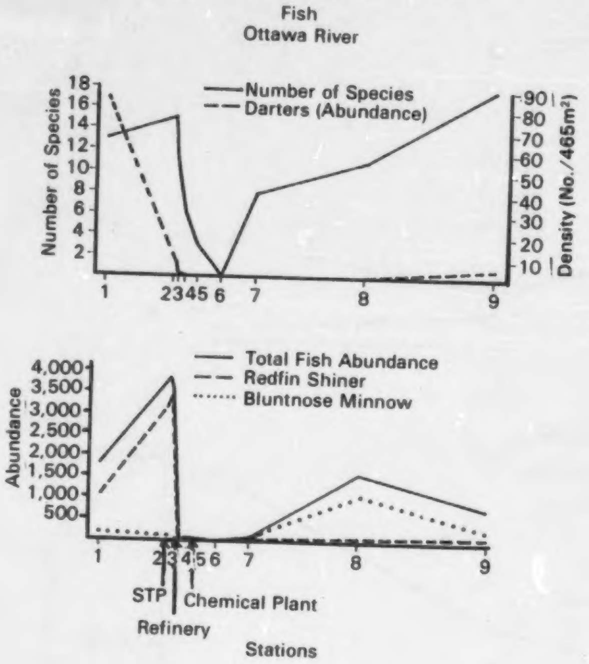
Figure 9-1. **Spatial distribution of fish community indices and associated parameters, 1982 survey.**



several stations, whereas no esocids were collected in the present study. Except for three species and one hybrid, all of the fish collected by EPA had been encountered previously in the Ottawa River by Ohio EPA or Trautman (1957) and Patrick et al. (1956) as summarized in Martin et al. (1979).

In 1976 and 1977, Station 1 was found to have a healthy community of fish. Although the grass pickerel, blackstripe topminnow, and the sucker family which were caught in 1976 and 1977, were not well represented in the present study, the abundance of other fish, especially the darters, indicates that Station 1 has not become more degraded since EPA's study. Station 2, sampled previously in 1977, was characterized as a stressed ecosystem by EPA because there was low species diversity and pollution-tolerant species were present. In the present study, a slightly better species diversity was found with a greater abundance of bluntnose minnow, redfin shiner, bluegill, and largemouth bass, plus two species of darter. Species composition at Station 4, located between the refinery and the chemical plant, was similar to that of the 1976 and 1977 studies with mainly fathead minnows, creek chubs, and green sunfish caught. Ohio EPA found the section of the Ottawa River between the chemical plant outfall and the Allentown dam to be "essentially devoid of fish populations" and located no stations in that section. EPA's Station 5, just downstream from the discharge of the chemical plant, comprised a poor fish community of only three species. No fish were caught at Station 6, located nearly 5 km downstream from Station 5.

Figure 9-2. Spatial distribution of selected fish species and community parameters, 1982 survey.



Station 7 showed low diversity during both the Ohio EPA and the present studies; however, carp, white suckers, and several sunfish predominated in 1977, whereas six species of minnow and the bluegill comprised the 1982 species list. As this station may represent early stages of recovery, the fish community may be in a constant state of flux as fish move upstream from more healthy sections.

The number of species has decreased from 14 to 10 at Station 8, and the number of families represented dropped from four to two. However, these data comparisons are not adequate to properly judge a change in the general health between 1977 and the present study. No darters were captured in either study.

In both studies, the number of species collected at Station 9 returned to levels found at the upstream control station, but with reduced numbers and diversity of darters. Ohio EPA considered the presence of the greenside darter, the logperch (*Percina caprodes*), and the blackside darter as demonstrating that marked improvement had occurred in water quality. One blackside and three greenside darters were caught (the greenside darter in an additional qualitative sample) at this station in 1982, suggesting no marked change in water quality since 1977 and recovery from upstream discharge effects is apparent at this station.

The condition of the Ottawa River just above the first of the three discharges appears to be healthy, based on the abundance of fish and number of species collected at Station 2. However, the reduced population of pollution-sensitive darters may indicate some degradation caused by the City of Lima or other point-source discharges. The effluent from the STP appeared to not substantially affect the fish community except for the virtual elimination of the darters. The effluent from the refinery apparently has had greater adverse effects upon the fish community in that the abundance and variety of fishes are greatly reduced at Stations 4 and 5 compared to that found upstream.

The total absence of fish at Station 6, located over 5 km downstream from the total discharge, suggests a delayed effect such as a high BOD. Lethally low dissolved oxygen levels could occur at night, even though acceptable levels of 5.7 and 6.8 mg/liter were recorded during benthos and fisheries collections, respectively, around midday.

It does not appear that the variable proportions of pool and riffle habitats among the stations can be correlated to the differences in fish communities. Major differences in the fish communities were found within each of two groups of stations where habitat proportions were similar. (These groups consisted of Stations 2, 4, 6, and 9 and Stations 3, 5, and 7). In addition, Station 1 contained the smallest percentage of riffle habitat, yet produced the greatest abundance and diversity of darters, which are generally a riffle-dwelling species. Conversely, no darters were caught at Station 8, which had a very large proportion of riffle habitat.

Recovery of the fish community appeared to occur in stages with distance downstream. The minnow component appeared to return to normal population levels (as illustrated by the reference stations) at Station 8 located approximately 34.5 km downstream of the refinery outfall, although the dominance of the minnows shifted from redfin shiners upstream of the refinery to bluntnose minnows below the refinery. The darters, on the other hand, did not reach recovery until Station 9 which is 58.5 km downstream from the refinery. Recovery of the darters was exemplified more by variety than abundance. However, the character of the habitat at this furthest downstream station was fairly dissimilar to that of the reference station because of greater flows and sources of input, and total recovery of darters to population levels observed at the reference station may not be possible because of their habitat preference for shallow-water riffle areas.

10. *Fish Caging Study, 1982 Survey*

An *in situ* caging experiment was carried out at six stations on the Ottawa River. Station 1 at Thayer Road was the reference station free from influence from Lima, and Station 2 before Sewage Treatment Plant (STP) was used to identify any effects from runoff and discharges from the City of Lima. One station was located in each of the three discharges examined: the STP (Station 3), the refinery (Station 4), and the chemical plant (Station 5). Station 9 at Kalida, Ohio, was located approximately 60 km downstream from the three discharges to observe recovery.

10.1 *In Situ* Toxicity Testing

The greatest mortalities occurred at the three discharge stations, where 50 percent of the test population or greater died after six days (Table 10-1). The refinery discharge was the most toxic, with only 30 percent surviving. The greatest rates of mortality at the discharge stations occurred after two days of exposure (Figure 10-1). As shown by the dye study conducted at the chemical plant (Figure 6-4), the cages at Station 5 were not within the discharge plume of the chemical plant. The mortality here possibly can be attributed to a diluted refinery discharge, or a combination of STP and refinery discharges.

The greatest 6-day survival (85 percent) was observed at Station 2, just upstream from the STP. A survival of 78 percent occurred at Station 9 (Kalida), indicating substantial recovery. The mortality rates at these two stations were fairly constant over the six days of the study.

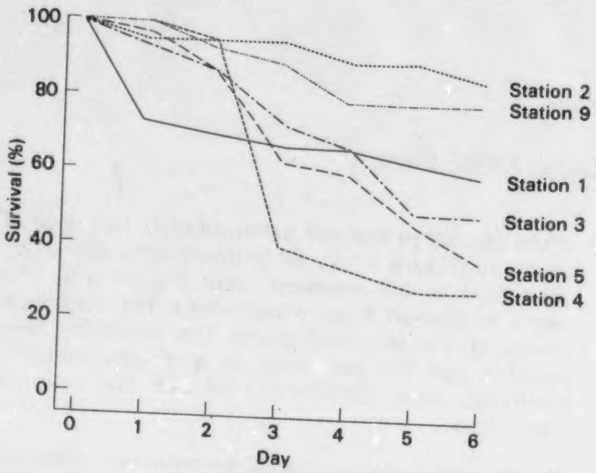
The poor survival at the upstream control station (Station 1, Thayer Road) raises some question as to the validity of this testing. One possible explanation

Table 10-1. Results of Fish Caging Study, Ottawa River, 1982 Survey

Station	Number of Fish Surviving per Cage						
	Day 0	Day 1	Day 2	Day 3	Day 4	Day 5	Day 6
1A	10	9	9	9	9	8	8
1B	10	7	6	5	5	5	5
1C	10	6	6	6	6	6	5
Total % Survival		73	70	67	67	63	60
2A	10	9	9	9	9	9	8
2B[a]	-	-	-	-	-	-	-
2C	10	10	10	10	9	9	9
Total % Survival		95	95	95	90	90	85
3A	10	10	9	8	6	6	6
3B	10	10	10	9	9	5	5
3C	10	8	7	5	5	4	4
Total % Survival		93	87	73	67	50	50
4A	10	10	20	4	4	4	4
4B	10	10	9	4	3	2	2
4C[a]	-	-	-	-	-	-	-
Total % Survival		*100*	*95*	*40*	*35*	*30*	*30*
5A	*10*	*10*	*9*	*7*	*6*	*5*	*4*
5B	*10*	*9*	*9*	*5*	*5*	*5*	*4*
5C	*10*	*10*	*8*	*7*	*7*	*4*	*3*
Total % Survival		*97*	*87*	*63*	*60*	*47*	*37*
9A	*10*	*10*	*8*	*7*	*5*	*5*	*5*
9B	*8*	*8*	*8*	*8*	*7*	*7*	*7*
9C	*10*	*10*	*10*	*10*	*10*	*10*	*10*
Total % Survival		100	93	89	79	79	79

[a] Missing cages.

Figure 10-1. Results of *in situ* fish caging study, Ottawa River, 1982 survey.



for the high mortality is that the increased stress of transportation in the holding tank may have eliminated the weaker fish and allowed the hardier individuals to be placed in cages at other stations. This is supported by the fact that the greatest rate of mortality at Station 1 occurred during the first day, after which it was similar to that at Stations 2 and 9. These results emphasize the need for further testing and development of these methods.

11. Benthic Macroinvertebrate Community, 1983 Survey

Both qualitative and quantitative collections were taken during the 1983 survey of the Ottawa River, thus increasing the number of habitats sampled at each station. Quantitative collections were taken in riffle areas as in the 1982 survey. Qualitative collections were taken along shore zones and pool areas.

11.1 Community Structure

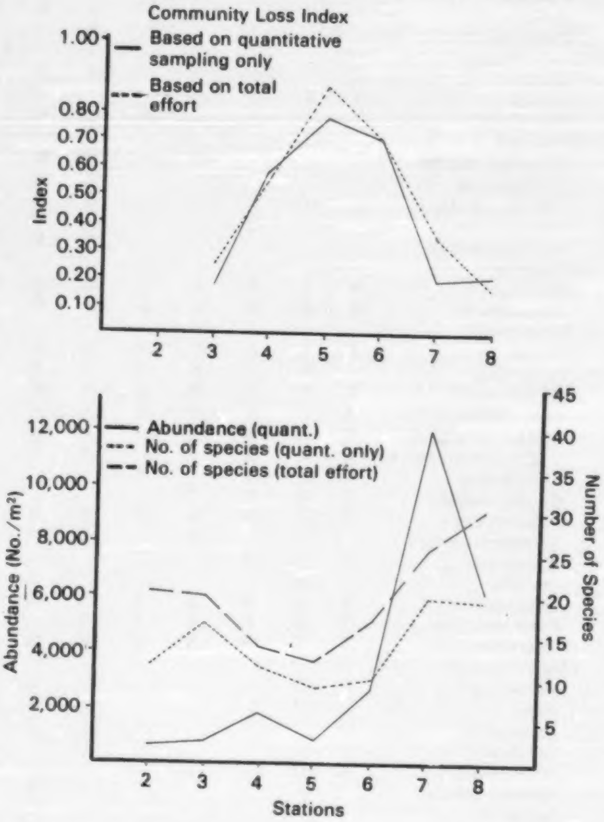
The number of taxa collected at each station ranged from 12 to 31 (Table 11-1). The largest variety of taxa were the chironomids which were represented at each station by several genera. The benthic community at Station 8 comprised the most taxa (31) of any station, having a greater variety of caddisflies, molluscs, and beetles than the other stations. The benthic community at Stations 4 and 5 was the least diverse, having relatively few taxa other than chironomid larvae.

Abundance distribution of the benthos exhibited somewhat different information on spatial trends than did composition. The benthic community was least abundant at Station 2 which is the reference station (Table 11-2). The greatest abundance was at Station 7, with over 12,000 organisms/m². The difference in abundances was primarily due to the simuliids and, to a lesser extent, chironomids.

The community loss index as described in Section 8.1 was calculated on the data obtained in the 1983 survey. Two separate indices were calculated and are based either on total taxa encountered in all sampling efforts (Table 11-1) or in the quantitative collections only (Table 11-2). The community loss index indicated that station dissimilarity to the reference station (Station 2 was used as the reference station of comparison in 1983 as opposed to Station 1 which was used in 1982) increased from a minimum at Station 3 to a maximum at Station 5, then decreased until Station 8 (Figure 11-1). Stations 4, 5, and 6 were the most dissimilar to Station 2 in composition. Very little difference in values was obtained when the qualitative sampling effort was included in the calculations.

The spatial trend illustrated by the community loss index was reflected by the trend in number of species (Figure 11-1). As the community loss index increased in value, the number of species decreased. The abundance of the benthos generally increased from Station 2 to peak densities at Station 7, then increased to the second highest abundance at Station 8. The substantial increase in numbers at Station 7, as mentioned previously, was due to the overwhelming dominance of simuliids (Table 11-2). The large density at Station 8 was attributed to the caddisflies.

Figure 11-1. Spatial trends of benthic community parameters, 1983.



11.2 Spatial Trends of Major Groups

The abundance of the major groups generally increased from upstream to downstream. The trichopterans (caddisflies) reached maximum densities at Stations 7 and 8 (Figure 11-2). The ephemeropterans

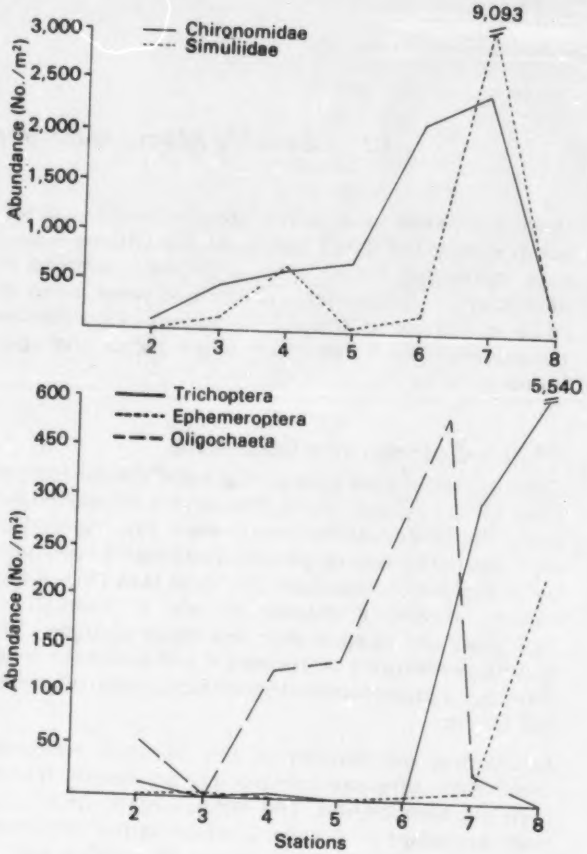
Table 11-1. Composition of the Benthic Community of the Ottawa River, July 1983

Taxa	Station 2	3	4	5	6	7	8
Coleoptera							
Psephenus L.	X						
Stenelmis L.	X	X	X	O	O	X	X
Stenelmis A.	X	O			X	O	X
Dytiscidae							O
Agabetes L.	X						
Dytiscus L.					X	X	
Laccophilus A.							X
Hydrophilidae L.		X			O		
Berosus L.				X	X	X	X
Berosus A.		X				X	X
Peltodytes A.						X	
Ephemeroptera							
Baetis	X					X	X
Caenis	X	X					X
Stenonema					X	X	X
Trichoptera							
Cheumatopsyche	X					O	X
Hydropsyche						X	X
Hydropsychidae P.						O	O
Hydroptilidae L.			O				O
Hydroptilidae P.						O	X
Simuliidae							
Simulium L.	X	X	X	X	X	X	X
Simulium P.	X	X	X	O	O	X	O
Chironomidae							
Pupae	X	X	X	X	X	X	O
Procladius		X		X	X	X	X
Ablabesmyia	X	X	X	X	X	X	X
Chironomus	O	X	X	X	X	X	X
C. (Dicrotendipes)	X	X	X			X	O
C. (Cryptochironomus)	X	X	X			X	X
C. (Tribelos)	X	X	X		X		
Glyptotendipes	X	X	X				
Polypedilum	X	X	X	X	X	X	X
Stictochironomus				X		O	X
Tanytarsus	X	X	X				
Zavrelia		O					
Cricotopus	X	X	X	X	X	X	X
Psectrocladius	O	O	O	X	X	X	
Oligochaeta	O	X	X	X	X	X	X
Miscellaneous							
Hetaerina	X	O					X
Argia		O				X	X
Turbellaria	O						O
Hirudinea					O	O	
Procambarus	X	O		X	X	X	X
Hyallela azteca						X	X
Corixidae							O
Gastropoda (snail)							X
Ancylidae							X
Hemerodromia L.	X	X	O	O		X	O
Hemerodromia P.	O	O	X	O		O	O
Ceratopogonidae L.							
Ceratopogonidae P.					X	O	X
Tabanidae L.						X	
Chaoborus					X		
Other							O
Total no. taxa[a] w/qual.	21	20	14	12	17	26	31
Community Loss Index (qual. & quant.)		0.25	0.57	1.00	0.70	0.35	0.16

[a]Multiple life stages, higher taxonomic levels, and Oligochaeta are not included in number of taxa.

Note: O = presence of species in quantitative samples only.
X = presence of species in qualitative samples (may include quantitative samples).

Figure 11-2. Spatial trend of major benthic taxonomic groups, 1983.



(mayflies) were present in low densities at Station 7, but were most abundant at Station 8. Neither of these two groups were abundant at the reference station (Station 2). The dipterans (represented by simuliids and chironomids) and oligochaetes were the principal benthic components of the community in the degraded area downstream of the discharges. The simuliids peaked at Stations 4 and 7 which were located downstream of sewage treatment plants and may have received nutrient enrichment from these sources. All of these groups declined in abundance at Station 8.

Two species each for the ephemeroptera (*Caenis* and *Baetis*) and Trichoptera (*Hydropsyche* and *Cheumatopsyche*) reflected the abundance trends for those groups (Table 11-2). The oligochaetes were not identified any further than class level so key species are not known for the 1983 survey. *Cricotopus* was the numerically dominant midge in 1983, but *Polypedilum, Ablabesmyia,* and *Chironomus* were also abundant at certain stations.

Table 11-2. **Abundance (No./m²) of Benthic Macroinvertebrates Collected from the Ottawa River, July 1983**

Taxa	Station						
	2	3	4	5	6	7	8
Coleoptera							
Psephenus L.	7.57						
Stenelmis L.	248.60	11.30	158.20	3.73	11.30	67.80	387.93
Stenelmis A.	64.07	3.73			7.57	15.03	184.53
Dytiscidae							3.73
Agabetes L.							
Dytiscus L.					3.73		
Laccophilus A.							
Hydrophilidae L.					3.73		
Berosus L.					3.73	3.73	3.73
Berosus A.							3.73
Peltodytes A.							
Total	320.24	15.03	158.20	3.73	30.06	86.56	583.65
Ephemeroptera							
Baetis						3.73	169.50
Caenis		3.73					
Stenonema					3.73	7.57	60.23
Total		3.73			3.73	11.30	229.73
Trichoptera							
Cheumatopsyche	*15.03*					*56.50*	485.9
Hydropsyche						210.41	4,810.07
Hydropsychidae P.						22.60	210.97
Hydroptilidae L.			3.73				7.57
Hydroptilidae P.						11.30	26.33
Total	15.03		3.73			300.81	5,540.84
Simuliidae							
Simulium L.		101.70	489.63	18.87	161.93	7,405.23	22.6
Simulium P.		45.20	192.1	7.57	18.87	1,687.43	7.57
Total		146.90	681.73	26.44	180.80	9,092.66	30.17
Chironomidae							
Pupae	15.03	26.33	22.6	64.07	116.73	229.73	3.73
Procladius				3.73		3.73	
Ablabesmyia	3.73	60.23	37.63	113.0	139.33	365.33	18.87
Chironomus	3.73	15.03	3.73	22.6	241.03	158.2	7.57
C. (Dicrotendipes)	7.57	11.3				3.73	3.73
C. (Cryptochironomus)			7.57				18.87
C. (Tribelos)		22.6	7.57				
Glyptotendipes	22.60	97.97	22.60				
Polypedilum	7.57	128.03	45.20	7.57	18.87	312.67	173.23
Stictochironomus						7.57	
Tanytarsus	7.57	15.03				3.73	7.57
Zaurelia		7.57					
Cricotopus	11.30	94.13	519.8	493.47	1,608.33	1,348.43	56.5
Psectrocladius	3.73	7.57	7.57	11.3	22.6		
Total	82.83	485.79	674.27	715.74	2,146.89	2,433.12	290.07
Oligochaeta	52.77		131.87	143.17	519.8	22.6	
Miscellaneous							
Hetaerina		3.73					
Argia	11.3						
Turbellaria	86.67						3.73
Hirudinea					7.57	3.73	
Procambarus	30.17	3.73				15.33	
Hyallela azteca						18.87	
Corixidae							3.73
Gastropoda (snail)							
Ancylidae							
Hemerodromia L.		56.5	30.17	3.73		11.30	3.73
Hemerodromia P.	11.30	7.57	22.60	15.03		18.87	3.73
Ceratopogonidae L.							
Ceratopogonidae P.						3.73	
Tabanidae L.							
Chaoborus							
Other							3.73
Total	128.14	82.83	52.77	18.76	7.57	71.83	18.65
Total densities	599.01	734.28	1,702.57	907.84	2,888.85	12,018.88	6,693.11
Total no. taxa[a]	13	17	12	9	10	20	19
Community Loss Index		0.18	0.58	0.78	0.70	0.20	0.21

[a] Multiple life stages, higher taxonomic levels, and Oligochaeta are not included in number of taxa.
Note: L. = larvae; A. = adults; P = pupae.

11.3 Comparison Between 1982 and 1983 Surveys

The level of identification between the two surveys was somewhat different so comparisons of composition and relative abundance are limited. The collection techniques for quantitative assessment were similar with the exception of the mesh size. The Hess sampler used in the 1982 survey was equipped with a 363-μm mesh screen, and the Hess used in 1983 had a 800-μm mesh screen. The larger mesh screen would not capture the early instars or small organisms encountered with the finer mesh.

Generally, the community composition of the benthos was similar between 1982 and 1983. The numerically dominant taxa during both surveys were simuliids, chironomids, oligochaetes, and, to a lesser extent, trichopterans and ephemeropterans. Spatial trends of community parameters (number of species, total abundance, community loss index) and major group densities were similar between the two surveys except for certain station shifts in peak densities. Station 2 was the only reference area sampled in 1983 and did not produce a large number of species or large abundances of sensitive species collected in 1982. Degradation of benthos was noted in both surveys to occur downstream of the three primary discharges with initial recovery occurring at Station 7. However, nutrient enrichment appeared to be extensive at Station 7 in 1983 resulting in peak densities of Simuliidae.

12. *Fish Community, 1983 Survey*

All available habitats were sampled at each of the seven biological stations using the same gear as in 1982. The proportional representation of each habitat to the total sampling area at each station is not available.

Table 12-1. Results of Fish Collections in the Ottawa River, July 1983

Species	Station 2	3	4	5	6	7	8	Total
Gizzard shad	4					2		6
White sucker	6							6
Golden shiner						4		4
Creek chub	15		4	1		4	56	80
Stoneroller	16					1		17
Fathead minnow	8	2	2		6	75	4	97
Bluntnose minnow	28	1	2			15	42	88
Spotfin shiner						186	85	271
Emerald shiner	2						2	4
Redfin shiner	511	130	14			17	3	675
Silverjaw minnow						7		7
Larval cyprinids		2		14			164	180
Rock bass	1							1
Largemouth bass	1							1
Green sunfish	2	4				6	2	14
Bluegill	6					17	2	25
Orangespotted sunfish	21					20		41
Pumpkinseed	1					5	1	7
White crappie						36		36
Larval centrarchids		2		12	3[a]			17
Greenside darter	5	1						6
Rainbow darter	1							1
Johnny darter							2	2
Black bullhead						1		1
Total organisms	628	142	22	27	9	396	363	1,587
Number of taxa	16	5	4	1	1	15	10	23
Community Loss Index		2.2	3.0	15.0	15.0	0.4	0.8	

[a] Captured by dip net.

12.1 Community Structure

Twenty-three species of fish were collected in the study area composing six families (Table 12-1). The greatest abundance and number of species were found at the reference station (Station 2). The redfin shiner dominated the community at Stations 2 through 4, disappeared from Stations 5 and 6, and then returned in small numbers at Stations 7 and 8. The spotfin shiner became the numerically dominant species of fish in the recovery community which was found to occur at Stations 7 and 8. This species was not found upstream of Station 7. Very few darters were found in the study area. The darters were most abundant at Station 2 but were also present at Stations 3 and 8.

The Community Loss Index was the highest at Stations 5 and 6, indicating that the greatest dissimilarity in composition to the reference station occurred at these two stations (Table 12-1). Stations 7 and 8 were most similar to Station 2 according to the calculated index. The index indicated that the communities at Stations 3 and 4 were marginally affected by the discharges.

Fish abundance trends followed closely that of the minnows with two species being particularly abundant (Figure 12-1). The redfin shiner accounted for over 80 percent of the fish at Stations 2 and 3. Although the spotfin shiner was the numerically dominant species of fish at Stations 7 and 8, it represented <50 percent of the fish.

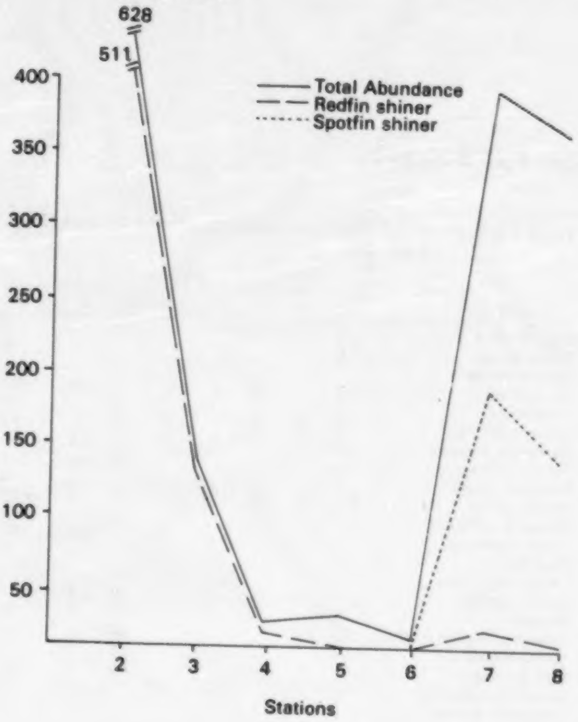
12.2 Comparison Between 1982 and 1983 Surveys

Compositional differences between the two surveys were subtle and probably due to differences in the level of effort and collection techniques. Abundances were lower in 1983 and may also be attributed to level of effort as well as natural seasonal fluctuation. However, numerical dominance among the stations was similar in both surveys with few exceptions.

Minnows dominated both surveys, but the recovery community in 1983 was dominated by spotfin shiners compared to bluntnose minnows in 1982. Bluntnose minnows were also abundant in the recovery community at Station 8 (Table 12-1).

Results of the 1983 survey suggested a more degraded fish community at Station 3 downstream of the Lima Sewage Treatment plant (STP) discharge than noted in 1982. However, one darter and the second highest abundance of redfin shiners were found at Station 3. The trend in recovery of the fish community was similar to that indicated by the 1982 survey.

Figure 12-1. Spatial trends of selected fish abundance, July 1983.



13. *Zooplankton Community, 1983 Survey*

Micro- and macrozooplankton were only collected during the 1983 survey using a Wisconsin stream net with a 80-μm mesh net. These results are not comparable to the drift results which emphasized macrozooplankton and macroinvertebrate components of the drift. Algae included in the plankton also were identified and enumerated.

13.1 Community Structure

Total density of planktonic organisms fluctuated from 1 organism or cell per liter to a maximum of 35 per liter (Table 13-1). The algal components of the plankton were dominated by the dinoflagellate, *Ceratium*. Rotifers, *Brachionus* in particular, were the most abundant component of the zooplankton. Unidentified copepods and cladocerans composed the crustaceans which were most abundant at Stations 6 and 7.

Generally, total plankton was least abundant at Stations 3 through 5 (Figure 13-1). Algae constituted nearly all of the plankton at Station 2 and represented over 50 percent of the density at Stations 2 through 4. Zooplankton increased from minimum levels at Station 2 and peaked at Station 6, then decreased to minimum levels at Station 8. *Brachionus* contributed the single highest density at Station 6.

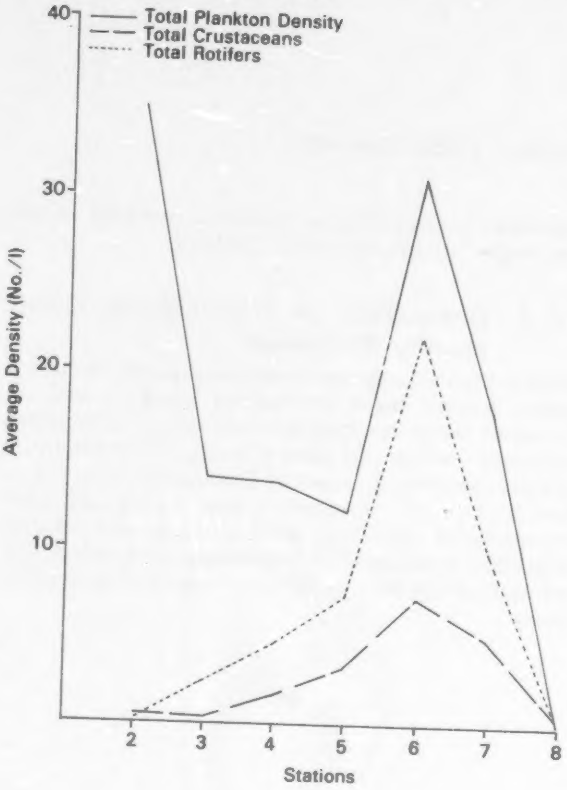
13.2 Evaluation of Zooplankton Community Response

Zooplankton abundance is relatively unimportant as a stable trophic level in riverine systems and its presence in low numbers at Stations 2 and 8 probably represent normal population levels. The substantial density increase observed at Station 6 may be due in part to nutrient enrichment from some upstream source, but is more likely attributed to a reduced level of grazing because of an absence of predators at the macroinvertebrate and plankton-feeding fish trophic levels.

Table 13-1. Planktonic Organisms (number/liter) Collected from the Ottawa River, July 1983

	Station						
	2	3	4	5	6	7	8
Crustaceans							
Copepods	0.07	0.14	0.72	0.68	0.48	0.88	0.05
Nauplii	0.44	0.40	1.0	1.50	1.52	1.49	0.26
Cladocerans	0.14	—	0.08	0.42	5.15	2.52	0.20
Rotifers							
Brachionus	—	0.07	2.48	6.7	21.92	9.6	0.05
Bdelloid rotifers	0.035	2.22	1.84	0.49	0.095	—	—
Keratella	0.27	0.13	0.28	0.16	—	—	—
Algae							
Ceratium	33.39	10.36	6.16	1.5	—	0.12	—
Desmids	0.34	0.14	0.04	0.1	1.46	3.09	0.46
Pediastrum	0.34	0.61	0.56	0.22	0.03	0.12	0.05
Other							
Chironomidae larvae	—	—	0.24	0.46	0.48	0.66	—
Nematoda	—	—	0.40	—	—	—	—
Total density	35.02	14.07	13.8	12.23	31.14	18.48	1.07
Total crustaceans	0.65	0.54	1.8	2.6	7.15	4.89	0.51
Total rotifers	0.30	2.42	4.6	7.35	22.02	9.6	0.05
Total algae	34.07	11.11	6.76	1.82	1.49	3.33	0.51

Figure 13-1. Spatial trends of zooplankton components of the plankton, July 1983.



14. Comparison of Laboratory Toxicity Data and Receiving Water Biological Impact

A primary objective of the Complex Effluent Toxicity Testing Program is to determine how effectively effluent toxicity testing predicts impact to the biota of the receiving system. The predictive capability of toxicity tests can be assessed by comparing effluent toxicity measurements (expressed as concentration-based effect levels) to actual instream biological impact (measured by standard biosurvey techniques). Dye studies determine plume configurations over the course of several days at a low flow period. Effluent toxicity concentrations are then compared to effluent concentration isopleths instream and biological impact zones. Where effect level concentrations are exceeded instream, biological impact is predicted. In this study, a direct correlation between measured effluent toxicity levels, instream concentrations, and adverse impact to the biota in the receiving water was considered a strong indication that measured effluent toxicity does measure instream degradation and can be translated directly into an assessment of adverse water quality impact.

In the development of permit limits, a quantifiable relationship must be established between an effluent and adverse impact to the local biota. Biosurveys are useful in identifying impact but are of lesser value in determining the amount of treatment needed to reduce that impact. Further, where several dischargers release wastewaters, impact assessment using biosurveys can become complicated and difficult to interpret.

Toxicity data, expressed as an effect concentration (such as a No Observable Effect Level or NOEL) can provide the quantification needed to set treatment requirements to reduce toxic water quality impact. If the NOEL is not exceeded instream, it can be concluded that no toxic impact will occur, assuming that bioaccumulation/human health is handled elsewhere and assuming that the proper NOEL is used.

A major difficulty in translating effluent toxicity to instream impact is the practical limitations the regulatory process places on data acquisition. Uncertainty in comparing laboratory toxicity data to instream impact arises when a limited amount of data are available on the toxicity of the effluent and the behavior of that effluent after discharge to the receiving water. Scientific certainty in the translation process is highest where a complete database is available for a discharge situation. A complete database would include acute and chronic toxicity data on a wide spectrum of indigenous species, ecosystem structure and function data, and daily exposure analysis over a long period of time. Unfortunately, ideal databases will be, practically speaking, nonexistent due to cost and analytical capability limitations.

Two principle sources of uncertainty in the translation process are species sensitivity and fate/persistency after discharge.

To be effective indicators of adverse impact, the species tested must be "sensitive" to the effluent's toxicity. If the test organisms were not representative of sensitive indigenous species in the ecosystem, the effluent could exert a toxic effect on some of the receiving water biota but not on the test organisms. A false negative analysis would result.

Different species exhibit different sensitivities to toxicants. There often are two orders of magnitude difference between the least sensitive and the most sensitive organisms when they are exposed to a particular toxicant or effluent. This range varies greatly and can be narrow or wide depending on the toxicant or effluent involved. The primary goal in toxicity analysis is to use a "sensitive" organism to test effluent toxicity. A majority of the biota exposed to that effluent in the receiving water will exhibit a lower sensitivity to that effluent and will be protected so long as that test organism's measured no-effect level is not exceeded. Since the measured toxicity of an effluent will be caused by unknown toxic constituents, the relative sensitivities of the test organisms will also be unknown. Therefore, proper effluent toxicity analysis requires an assessment of a "range" of sensitivities of test organisms to that effluent. The only way to assess sensitivity range is to test a number of different species.

In this study, two organisms were used to assess a range of sensitivity to the effluent toxicities of the three dischargers. They exhibited different responses to the different effluents. The test organism exhibiting sensitivity at the lowest concentration for that effluent was considered representative of the sensitive organisms in the receiving water.

Effluent toxicity fate/persistence must be considered in making the translation between laboratory toxicity tests and adverse impact. As soon as an effluent mixes with receiving water, its properties begin to change. The rate of change of toxicity is a measure of the persistence. In most cases, the level of toxicity instream will drop as decay processes (photodecomposition, microbial degradation) or compartmentalization processes (sediment deposition, volatilization) occur and bioavailability decreases.

If the toxicity measured in laboratory toxicity tests is quickly reduced or eliminated after discharge, the translation between toxicity and impact will not be valid.

Onsite toxicity testing from this study and other subsequent studies has indicated that the toxicants causing toxicity measured at discharge sites tend to be persistent. There is little "near field" degradation of the measured effluent toxicity. Effluent toxicity does exhibit "far field" decay. Typical patterns of progressive downstream decreasing toxicity (similar to BOD decay) have been observed in a number of discharge situations. In this study, ambient toxicity test data were used to assess the fate/persistency of measured effluent toxicity.

With these two sources of uncertainty taken into consideration, the analysis of the effectiveness of effluent toxicity tests to measure actual instream impact was conducted.

14.1 Results of Integration Analyses

Regression analyses were performed using the results of ambient toxicity tests and aquatic community measures. The analyses indicate there is a correlation between ambient chronic toxicity and the number of species, community loss, and diversity of the aquatic invertebrates and algae (Table 14-1). A positive relationship existed between young production of *Ceriodaphnia* and number of benthic species (R=0.71) and benthic diversity (R=0.79), while a negative correlation (R=0.63) was observed between young production and benthic community loss (Figure 14-1). The good correlation between young production of *Ceriodaphnia* and benthic parameters was a result of an agreement in effects from the STP. The number of benthic species decreased from 52 at the reference station to 32 at Station 3, and a similar reduction in diversity (from 3.6 to 1.7) occurred between the reference station and Station 3 (see Chapter 8). Young production for *Ceriodaphnia* did not occur in tests on water collected from this station (see Chapter 4) where the effluent from the STP constituted over 50 percent of the flow (Table 6-2).

Table 14-1. Comparison of Toxicity and Biological Response

Station	Ambient Toxicity Impairment (%)	Biological Community Impairment (%)
	1982 Data	
2	0	0
3	100	40
4	52	60
5	63	80
6	19	100
7	0	40
8	0	16
	1983 Data	
2	0	0
3	91	70
4	59	75
5	71	80
6	86	94
7	77	7
8	5	40

These data from the benthic survey and the *Ceriodaphnia* testing indicate a severe chronic condition below the STP which was attributable to that effluent.

The algal community also decreased in diversity below the STP (see Chapter 7) which resulted in a high correlation with young production of *Ceriodaphnia* (Figure 14-2). The algal community appeared to improve following the refinery discharge. However, this apparent amelioration of effects was temporary and was attributable to an influx of algae from the refinery waste treatment pond.

No correlation resulted between *Ceriodaphnia* fecundity and fish species and diversity. Effects on the fish community between the STP and refinery were slight (see Chapter 9).

Results of the effluent toxicity testing on larval fathead minnows indicated that the STP did not exhibit chronic toxicity to the fathead minnows. The differences in response exhibited by the chronic effects upon the fecundity of *Ceriodaphnia* and the growth of larval fathead minnows indicate the need to include measurements of chronic toxicity of the effluents for as many species as possible.

The effluent toxicities measured by the effluent toxicity tests exhibited a high level of persistence. This property can be verified by the analysis of the ambient toxicity data. The ambient chronic toxicity in the Ottawa River was reflected in the observed impact from the field studies (Figures 14-3 and 14-4). Chronic toxicity was measured at Stations 3, 4, 5, and 6 (a distance of 14.5 km). It was at these stations that the invertebrates, fish, and algae were impacted most heavily. For the invertebrates and algae, the relationship (as shown in Figures 14-1 and 14-2), is linear. At Station 7, no chronic toxicity was observed in *Ceriodaphnia* toxicity tests. This was the first station where the benthos and algae began to show recovery.

Figure 14-1. Correlation of *Ceriodaphnia* young per female with benthic parameters from eight stations in the Ottawa River, Lima, Ohio, 1982.

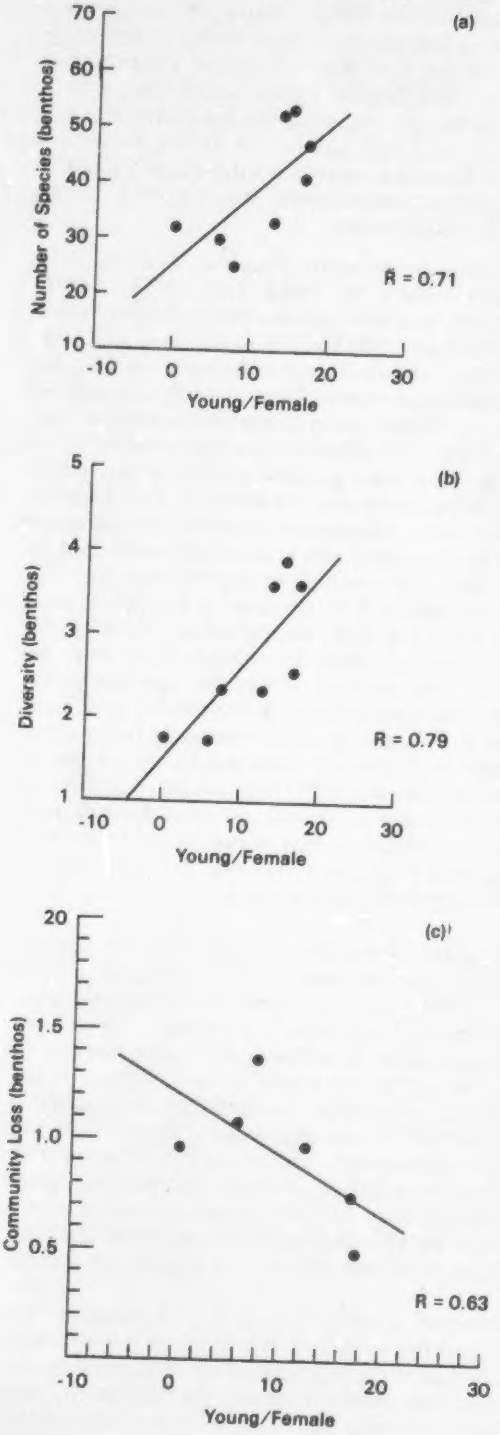


Figure 14-2. Correlation of *Ceriodaphnia* young per female and algal diversity at eight stations in the Ottawa River, Lima, Ohio, 1982.

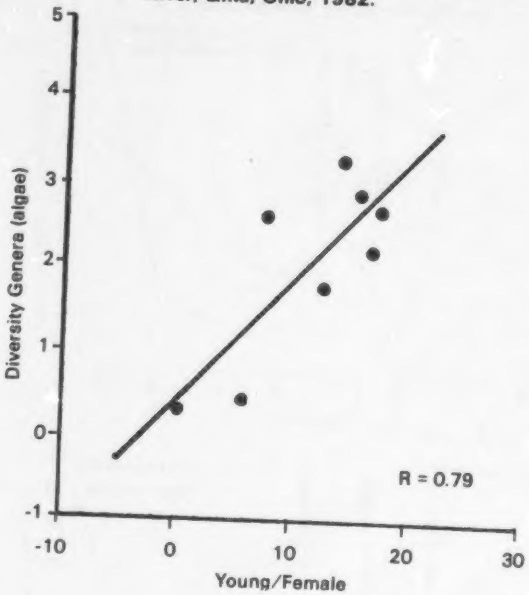


Figure 14-3. Ambient toxicity correlation between *Ceriodaphnia* young per female and ecological survey data for 1982.

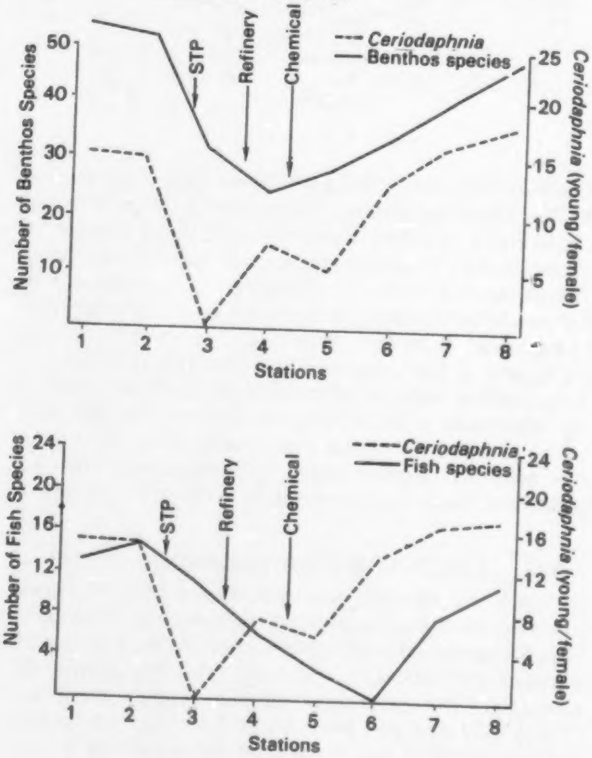
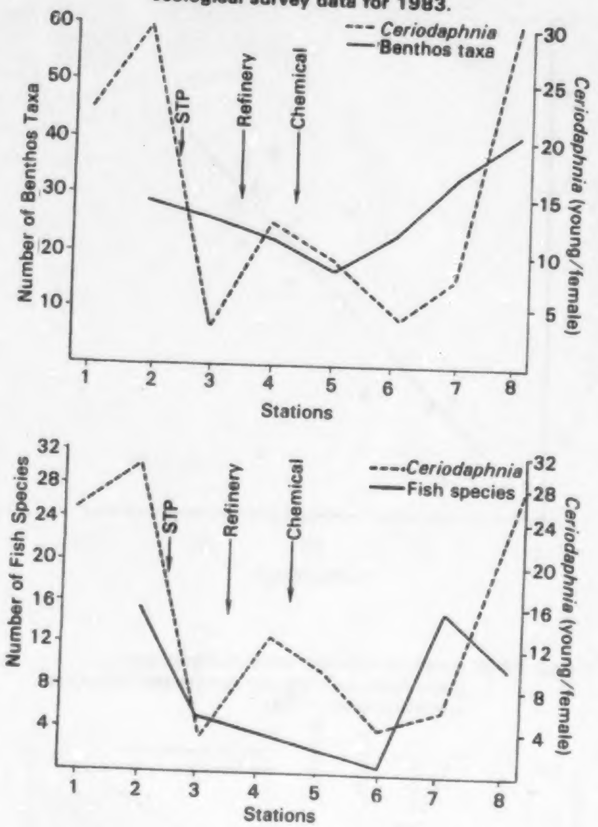


Figure 14-4. Ambient toxicity correlation between *Ceriodaphnia* young per female and ecological survey data for 1983.



The fish community did not recover fully until Station 9. The cause of delay in recovery is unknown, but could have resulted from low dissolved oxygen or other sources of pollutants, such as landfill leachates. The difference in the sensitivity of *Ceriodaphnia* and fish could be the reason for the degradation of the fish community between Stations 7 and 9 was not predicted by *Ceriodaphnia* toxicity. The relationship between the fathead minnow test results and the fish survey needs to be strengthened with further tests using fathead minnows in ambient tests. However, the effluent dilution tests with fathead minnows agree well with the impact on the fish community.

14.2 1982-1983 Comparison

In 1982 the benthic population was most severely impacted in the zone where the *Ceriodaphnia* toxicity was observed. At stations where there was up to a 50 percent reduction in *Ceriodaphnia* fecundity, 50 percent of the macroinvertebrate species were lost (Table 14-1, Figures 14-1, 14-3). Likewise, when less chronic toxicity was measured, as indicated by the production of young *Ceriodaphnia*, the benthic diversity increased. At Station 8 where the periphyton and benthos had fully recovered, there was no ambient toxicity measured. In 1983, the toxicity and impact patterns of the Ottawa River from Stations 2 through 8 were quite similar. The number of benthic taxa identified in 1983 (Figure 14-4) is not as large as in 1982 (Figure 14-3) because all the taxonomy was not completed in 1983 (see Section 11.3). The maximum toxicity was noted between Stations 3 and 6 and the maximum biological community impairment occurred at Station 5 in both years.

The fish community from Stations 2 through 8 responded similarly in 1982 and 1983. Subtle differences in the spatial trends of the fish community between 1982 and 1983 are probably due to differences in level of effort and collection techniques between the two years (see Section 12.2). It should be noted that a different group of chemicals were identified in the 1983 STP effluent from that present in the 1982 study. However, general effects at Station 3 were similar between the two years for both the fish and invertebrates. Maximum impairment took place at Station 6 where only one species was collected. In 1982 no fish were collected at this station; the problem at Station 6 is unclear, although it was suggested from the 1982 ecological survey (Chapter 9) that a delayed effect from high BOD may be responsible. The number of benthic species (taxa) increased at Station 6 from the immediate upstream stations in both years. In 1983, however, the toxicity impairment at Station 6 was nearly as great as downstream from the STP. This would indicate the possible presence of additional toxicants. Subsequent investigation indicates that there is an industrial landfill adjacent to the river and seepage to the Ottawa River may be taking place.

The relationship between *Ceriodaphnia* toxicity response and biological impairment is quite good during each year. When such correlations exist as demonstrated with the 1982 data, the reduction in toxicity could be predicted to achieve a certain degree of recovery. However, for some groups of organisms, the pattern is not so clear. The lack of good correlation could result from the displacement of ambient toxicity caused by the variable nature of the effluent and stream flow. In addition, members of the biological community such as fish can move in response to a gradient or be affected by other stresses such as unfavorable dissolved oxygen or temperature levels.

Differences in toxicity response of laboratory test animals and the variation of biological impairments require that impacts in any community component be considered important. In analyzing the toxicity test data, toxicity to the greatest impairment indicated by *Ceriodaphnia* or fathead minnows, reproduction of

Ceriodaphnia, or the growth of newly hatched fathead minnows was used. Community impairment was based on that portion of the community that lost the highest percentage of species.

A method of comparing ambient stream toxicity and biological community impairment was developed to relate toxicity testing information to ambient community response (Table 14-1). The expected ambient stream toxicity was calculated by determining the minimum NOEL concentration of the effluent, regardless of test species. The biological community impairment was calculated by the maximum percent of species lost (relative to a reference area) regardless of trophic level. In 1982 downstream from the STP, there was a 100 percent toxicity impairment and only 40 percent community impairment, whereas in 1983 there was a 91 percent toxicity impairment and a 70 percent community impairment. This yearly difference was due to the fact that in 1982, only the benthic invertebrates and attached algae were impacted, whereas in 1983 the fish community was severely impacted. Downstream from the refinery at Station 4, in 1982 there was 52 percent toxicity impairment and a 60 percent biological community impairment (Table 14-1). In 1983, there was a 59 percent toxicity and a 75 percent community impairment. The similarity of ambient toxicity and community impairment in this reach indicates the ability to quantify chronic stream effects.

The results downstream from the chemical plant are the most difficult to interpret because of the upstream toxicity. In 1982 the impact measured in the chemical plant effluent by the toxicity test was from upstream water. Likewise, the toxicity measured in the stream and resultant biological impact probably resulted from upstream sources. The ambient toxicity indicated a 63 percent reduction in reproduction, while the measured biological impact was 80 percent. In 1983 the ambient toxicity measured caused an 88 percent reduction in reproduction from upstream sources. The biological community was reduced 80 percent. Thus, the agreement between ambient toxicity and community response is good.

In the recovery reach at Stations 7 through 8, the agreement between predicted ambient toxicity impairment and observed biological impairment is generally quite good. However, because of the transitory nature of the reach for both toxicity and measurement of organisms, there are some stations where both impairments are not equal. In the reach where full recovery of the community is achieved, toxicity impairment was not present.

In summary, there is a high correlation between ambient stream toxicity and the number of species, diversity, and community loss of aquatic invertebrates. The correlation between *Ceriodaphnia* and fish species at the same station was poor. Responses of the algal community correlates well with the *Ceriodaphnia* toxicity tests. Persistence of effluent toxicity can be measured using ambient stream toxicity. Use of more than one test species improves prediction of biological impact. In addition, many segments of the aquatic community are required in the assessment of biological impact. Biological community impairment occurred to the same degree as toxicity impairment. Where no toxicity was measured from upstream effluents, some impairment of the fish community was evident but none on the periphyton and benthos. Both *Ceriodaphnia* and fathead minnow chronic tests can be used to measure ambient toxicity. These data appear to be directly relatable to biological community impairment.

14.3 Calculation of Toxicity Reduction

Results of the effluent dilution tests analyzed together with the calculated effluent concentrations in the stream (determined by the dye dilution studies) allows further comparisons to be made. The 7-day fathead minnow chronic tests were conducted on the three effluents. The STP effluent at 100 percent concentration had no impact on growth of fathead minnows, and the number of fish species (11) below the STP outfall remained relatively high. Refinery waste of 50 percent caused nearly 90 percent mortality of fathead minnows in the effluent toxicity tests. The stream below the refinery was approximately 30 percent refinery waste. Only six fish species were found in this area. The number of fish species continued to decrease at the next three downstream stations below the chemical plant. The effluent dilution tests on the chemical plant effluent showed that the 100 percent effluent had no effect on fathead minnows. There was high mortality in the controls and lower concentrations of waste which contained upstream dilution water contaminated with about 30 percent refinery effluent. These concentrations were clearly toxic and thus agreed with the expected toxic impact which was observed in the reduced fish population in the stream below the refinery.

The refinery effluent at 10 percent concentration reduced fathead minnow survival 25 percent and at 50 percent concentration, growth was reduced about 60 percent. The Ottawa River downstream from the refinery outfall is about 29 percent refinery wastes. Using 10 percent refinery effluent as the no effect concentration for fathead minnows and 29 percent as the river concentrations of refinery effluent, a 2.9-fold reduction in toxicity would be necessary. The *Ceriodaphnia* young production was reduced 90 percent in all concentrations of the refinery effluent as the result of the dilution water which contained about 60 percent STP effluent. The no effect concentration of STP effluent for *Ceriodaphnia* was between

5 and 10 percent. If the concentration of 10 percent STP effluent is used as the no effect concentration, and considering that 77 percent of the Ottawa River below the STP outfall is STP effluent, an approximate 7.7-fold reduction in toxicity would be required to protect the community below the outfall at the flow during the study. For both the STP and refinery wastes, additional tests with smaller concentration intervals would make the estimates of needed reductions more precise.

The permissible loading must maintain a concentration less than or equal to the no effect concentration at the critical flow specified by the regulatory agency. For the Ottawa River, the permit limits are based on the 7-day low flow in 10 years (7Q10). If one calculates the amount of toxicity reduction necessary to remove chronic toxicity in the receiving stream at the time of study, it must then be extrapolated to the flow upon which the permit is based. The methods for calculating the required abatement can be based on ambient stream toxicity or effluent toxicity. There are two approaches to calculating the required abatement: the first is an engineering approach based on Toxic Units (Toxic Unit Approach); the second is a more biological approach which is based on stream and effluent flow and the no effect concentration (Concentration Approach).

The Toxic Unit Approach for calculating the required abatement can be based on ambient stream toxic units or effluent toxic units (TU). A toxic unit is the inverse of an effluent concentration producing a defined endpoint, e.g., if the no effect concentration is 10 percent, one would have 10 toxic units; for a 5 percent no effect concentration, 20 toxic units; for a 100 percent no effect concentration, one toxic unit. The following data are used in the examples of calculating abatement for the two approaches with the STP and refinery data. Examples assuming additivity of multiple effluents are also given.

Since the toxic units used here are based on cronic toxicity, toxic units are designated as TU_c.

Flows (m^3/sec):

7Q10 of STP	STP	Flow Upstream of Refinery	Refinery	Flow Upstream of Chemical Plant	Chemical Plant
0.071	0.36	0.43	0.20	0.62	0.055

No Effect and Effluent Toxic Units Data

Discharger	No Effect Concentration	Toxic Units (TU_c)
STP	10%	10
Refinery	3%	33.3
Chemical Plant	100%	1

For the following examples, assume that the river upstream of the STP has 0.0 TU_c.

Toxic Unit Approach

I. Sewage Treatment Plant Reduction Calculation

A. Ambient Stream Toxicity Balance Method:

For a single discharge, the equation is as follows:

$$\text{Ambient } TU_c = \frac{\text{Effluent Flow} \times \text{Effluent } TU_c + \text{Upstream } TU_c \times \text{Stream Flow}}{\text{Effluent Flow} + \text{Stream Flow}} \qquad (1)$$

Thus using Equation 1, ambient TU_c below the STP can be calculated.

$$= \frac{0.36 \times 10 + 0.0 \times 0.071}{0.36 + 0.071}$$

$$= 8.35$$

[This is 7.35 TU_c more (8.35-1.0) than permissible in the stream.]

In that the permissible stream TU_c is 1.0, one can calculate the reduction required as follows:

$$\text{Reduction Required (\%)} = \frac{\text{Ambient } TU_c - \text{Permissible Stream } TU_c}{\text{Ambient } TU_c} 100 \qquad (2)$$

$$= \frac{8.35 - 1.0}{8.35} \times 100$$

$$= 88$$

B. Effluent Toxicity Calculation Method:

The required abatement to achieve a no effect level of 1.0 TU_c in the stream is calculated:

$$1.0 = \frac{\text{Permissible } TU_c \times \text{Effluent Flow}}{\text{Stream Flow} + \text{Effluent Flow}} \qquad (3)$$

$$1.0 = \frac{\text{Permissible } TU_c \times 0.36}{0.071 + 0.36}$$

Solving for permissible TU_c,

Permissible TU_c = 1.2 in STP effluent.

Required abatement based on TU_c is,

$$\text{Reduction Required (\%)} = \frac{\text{Effluent } TU_c - \text{Permissible } TU_c}{\text{Effluent } TU_c} \times 100 \qquad (4)$$

$$= \frac{10 - 1.2}{10} \times 100$$

$$= 88$$

It should be noted that the reductions for the ambient stream balance method and the STP effluent toxicity method are the same (88%) because the upstream TU_c are assumed to be 0.0.

II. Refinery Reduction Calculation

A. *Ambient Stream Toxicity Balance Method:*

For this example, assume there is no additivity of TU_c among discharges. That is, upstream of the refinery, the ambient TU_c is 0.0.

$$\text{Ambient } TU_c = \frac{0.2 \times 33.3 + 0.0 \times 0.43}{0.20 + 0.43} \qquad (1)$$

$$= 10.57$$

$$\text{Reduction Required (\%)} = \frac{10.57 - 1.0}{10.57} \times 100 \qquad (2)$$

$$= 91$$

B. *Effluent Toxicity Method:*

Again, the permissible TU_c in the stream below the refinery is 1.0.

$$1.0 = \frac{\text{Permissible TU} \times 0.20}{0.43 + 0.20} \qquad (3)$$

Permissible TU_c = 3.2 in Refinery Effluent

Required abatement based on TU_c is thus,

$$\text{Reduction Required (\%)} = \frac{33.3 - 3.2}{33.3} \times 100 \qquad (4)$$

$$= 90$$

Concentration Approach

This approach utilizes biological effects concentrations such as the NOEL for calculating the reduction required rather than the engineering terms of the Toxics Unit Approach.

To avoid impairment in the stream, the instream waste concentrations (IWC), which is the effluent concentration in the stream after complete mixing, must be equal to or less than the NOEL[*] as determined in the effluent dilution test. Conversely, for a given waste concentration in the stream, the NOEL must be a percent waste that is equal to or greater than the IWC. Further, for determination of treatment requirements, the IWC must be less than or equal to the NOEL at the critical low flow as specified in the applicable standards.

I. Sewage Treatment Plant Reduction Calculation

For the Ottawa River the critical flow is specified as the 7Q10. During the study period the flow was 0.071 m^3/sec (~7Q10) above the STP outfall. Thus,

$$\text{IWC (\%)} = \frac{\text{Effluent Flow}}{\text{Effluent + Stream Flow}} \times 100 \qquad (5)$$

$$= \frac{0.36}{0.36 + 0.071} \times 100$$

$$= 84$$

The NOEL was approximately 10 percent for the STP effluent using the *Ceriodaphnia* test, so the required reduction is:

$$\text{Reduction Required (\%)} = \frac{\text{IWC - NOEL}}{\text{IWC}} \times 100 \qquad (6)$$

$$= \frac{84 - 10}{84} \times 100$$

$$= 88$$

The lowest permissible NOEL of the waste must equal the IWC which for the STP is 84%.

To determine the toxicity reduction needed for a critical flow different from the ones used in the example above, substitute that flow into equations 5 and 6 and solve. (For further explanation see footnote [*].)

II. Refinery Reduction Calculation

The data gathered in the 1982 study do not permit any similar calculations for the reduction required for the refinery because the dilution water for the refinery

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[*] Occasionally, the NOEL is calculated as the geometric mean of the lowest concentration that produces an effect and the highest concentration that does not produce an effect; but for large concentration intervals as used here, a better estimate is the lower of the two values.

[*] One can show that Equation 6 can be derived using Equation 4 as follows:

$$\text{Reduction Required (\%)} = \frac{\text{Effluent } TU_c - \text{Permissible } TU_c}{\text{Effluent } TU_c} \times 100 \qquad (4)$$

Where,

$$\text{Effluent } TU_c = \frac{1}{\text{NOEL}}$$

$$\text{Permissible TU} = \frac{1}{\text{IWC}} \quad \text{(for ambient } TU_c \text{ set at 1.0)}$$

By substituting into Equation 4,

$$\text{Reduction Required (\%)} = \frac{\frac{1}{\text{NOEL}} - \frac{1}{\text{IWC}}}{\frac{1}{\text{NOEL}}} \times 100$$

Then simplifying,

$$\text{Reduction Required (\%)} = \frac{\text{IWC - NOEL}}{\text{IWC}} \times 100$$

test was toxic due to the STP effluent contained in it. In the 1983 study, the refinery effluent was diluted with water not containing STP effluent. In that test, the NOEL was between 10 and 3 percent concentrations. Using 3 percent as the NOEL and the 1982 flow of 0.43 m^3/sec:

$$\text{IWC \%} = \frac{0.20}{0.29 + 0.43} \times 100 \quad (5)$$

$$= 32$$

$$\text{Reduction Required (\%)} = \frac{32 - 3}{32} \times 100 \quad (6)$$

$$= 91$$

The lowest permissible NOEL of the waste must equal the IWC which for the refinery is 32%.

This calculation assumes that the STP concentration in the upstream water is at or below the NOEL and that there is no additivity to the toxicity of the refinery waste by the STP effluent present. If discharge limits were to be developed, the best approach would likely be to permit no more than the NOEL of each one. The protectiveness of these concentrations could be tested by adding a concentration of each effluent equal to the NOEL to upstream water to see if jointly, any impairment would occur. If it did, then a next step would be to try dilutions of the mixture of the two NOELS in order to determine where there is no impairment. These then would become the permissible IWC for the critical flow.

In the examples above, the toxic effects of the wastes have been assumed to be non-additive. From the Ottawa River and other studies, wastes have been found to be generally non-additive. On occasion when the discharges were from the same type of industry (i.e., metal finishing), the toxic effects were additive. In no study was synergism found. In some studies one effluent rendered another effluent less toxic even though they were both toxic before mixing.

Multiple Effluents - Additivity

For multiple effluents that are believed to be additive, and when allocation is necessary, then the policies of the regulating agencies should be followed. There are many possible allocation approaches: (1) upstream design flow, (2) equal degree of treatment, (3) equal loading, (4) equal reduction. There could be other bases for allocation, all of which are based on policy rather than science. The approach used for dissolved oxygen waste load allocation could be applied most directly. Examples of the biological and engineering approaches are presented to illustrate the simplicity of calculation.

Toxic Unit Approach—Additivity

The ambient TU_c as calculated for two or more effluents is as follows:

$$\text{Ambient } TU_c = \frac{\sum_{i=I}^{n} \left(\text{Effluent Flow}_i \times \text{Effluent } TU_{ci}\right) + \sum_{j=I}^{m} \left(\text{Additive Stream Flow}_j + \text{Stream } TU_{cj}\right)}{\sum_{i=I}^{n} \text{Effluent Flow}_i + \sum_{j=I}^{m} \text{Additive Stream Flow}} \quad (4)$$

Where, n = number of effluents
m = number of upstream and tributary flows

And, additive stream flow includes upstream, upstream effluent and tributary flows.

The above equation can be used similarly to Equation 1 to calculate the ambient TU_c and the permissible TU_c. To solve for permissible TU_c, one must start with the upstream discharge and work downstream.

The equation for the STP would be the same as before since it is the most upstream discharge.

To calculate permissible TU_c in the downstream effluents such as for the refinery, the above equation can be simplified because the ambient TU_c below the STP (which is above the refinery) is set at 1.0 and likewise the ambient TU_c downstream of the refinery is set at 1.0.

The permissible TU_c for refinery is calculated:

$$1.0 = \frac{0.20 \times \text{Permissible } TU_c + 0.431 \times 1.0}{0.20 + 0.431} \quad (7)$$

Permissible TU_c = 1.0 in Refinery Effluent

$$\text{Reduction Required (\%)} = \frac{33.3 - 1.0}{33.3} \times 100 \quad (4)$$

$$= 97$$

In the above allocation procedure the benefits of having no toxic units in the upstream flow is only realized by the STP and not by the refinery.

If the allocation of the upstream flow is shared equally (0.071/2 = 0.0355) the required abatement is calculated:

For the STP:

$$1.0 = \frac{0.36 \times \text{Permissible } TU_c}{0.36 + 0.0355} \quad (7)$$

Permissible TU_c = 1.1 in STP Effluent

$$\text{Reduction Required (\%)} = \frac{10 - 1.1}{10} \times 100 \quad (3)$$

$$= 89$$

And, for the refinery:

$$1.0 = \frac{0.36 \times 1.1 + 0.20 \times \text{Permissible TU}_c + 0.0355 \times 0.0}{0.20 + 0.431} \qquad (7)$$

Permissible TU_c = 1/18 in Refinery Effluent (which compares to 1.0 when dilution was not equally shared)

$$\text{Reduction Required (\%)} = \frac{33.3 - 1.18}{33.3} \times 100 \qquad (3)$$

$$= 96$$

The differences for this additivity example are small because of the small amount of dilution flow compared to effluent flow.

Concentration Approach—Additivity

When additivity is assumed using the Concentration Approach, the NOEL should be reduced for each one. If this is done by lowering each one equally the calculation would then be for the STP:

$$\text{NOEL} = \frac{10}{2}$$

$$= 5$$

$$\text{Reduction Required (\%)} = \frac{84 - 5}{84} \times 100 \qquad (6)$$

$$= 94$$

$$\text{Permissible Toxicity (\%)} = \frac{84}{2}$$

$$= 42$$

And, for the refinery:

$$\text{NOEL} = \frac{3}{2}$$

$$= 1.5$$

$$\text{Reduction Required (\%)} = \frac{32 - 1.5}{32} \times 100 \qquad (6)$$

$$= 95$$

$$\text{Permissible Toxicity (\%)} = \frac{32}{2}$$

$$= 16$$

These values for the STP and the refinery, 94 and 95 percent, respectively, compare to 88 and 91 percent under the no additivity assumption. Obviously, other scenarios can be done comparably. The STP could be given two-thirds of the dilution capacity and the refinery one-third. All that is needed to calculate the reduction required is to multiply the STP NOEL by two-thirds and the refinery NOEL by one-third and recalculate Equation 6 for each one. Reductions can be calculated based on partial additivity. For example, 1.5 NOEL could be allowed below the refinery. If the division of dilution is to be equally proportioned between both dischargers, then 0.75 of each NOEL would be substituted in Equation 6. Three or more discharges are handled identically.

A slightly different case occurs if, for example, the refinery decides to reduce toxicity by reducing effluent flow. In this instance, Equation 5 must be solved for the new expected flow before Equation 6 is solved. In this example, the upstream discharge, being a STP, would not likely be able to reduce flow, but where the upstream discharge does reduce effluent flow, the reduction required of downstream discharges will be changed because total flow (effluent and stream flow) will change according to Equation 5.

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Appendix A.
Toxicity Test Methods

A.1 1982 Methods

For the effluent dilution (ED) tests, a grab sample of stream water was collected from just upstream of each outfall in the afternoon of the day before it was used. The effluent was collected as a 24-hour composite sample by continuously pumping approximately 10 ml/min from the discharge flow. Each daily composite was begun around 0800-0900 hours. All discharges were relatively constant, so the composite was essentially flow-proportional.

Dilution water was warmed to room temperature overnight and effluent samples were warmed on a hot plate to room temperature when test solutions were made for the ED tests. Samples for the ambient test were warmed as soon as they were brought to the lab and the animals transferred on the day of sample collection.

The various concentrations were made by measuring effluent and stream water using graduated cylinders of various sizes and mixing each concentration in a 3.8-liter polyethylene jar. Enough was mixed at one time for both the fathead minnow and *Ceriodaphnia* test. All samples were at or near dissolved oxygen (DO) saturation when solutions were made up except for Stations 4A through 7 of the ambient test. No chemical measurements for specific chemicals were made. Routine water chemistry such as DO and pH was measured in various samples daily, and many of the DO measurements were made just before changing test solutions to determine the minimum values occurring.

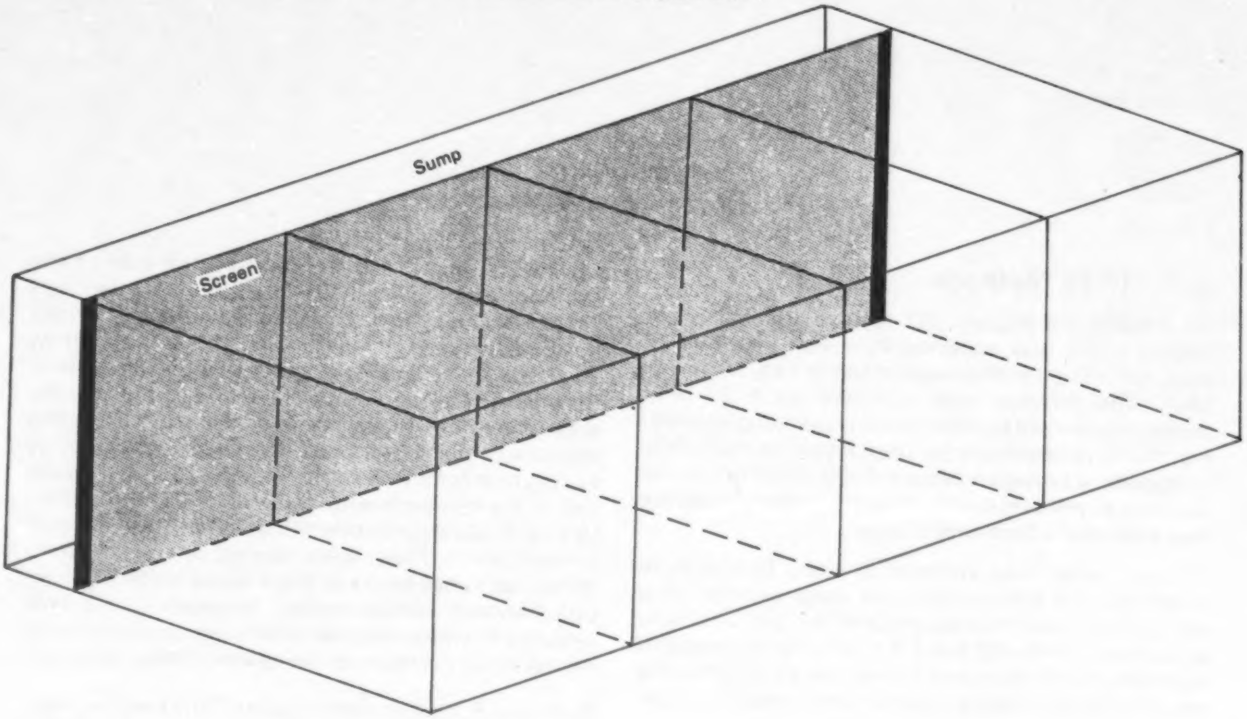
Test solutions were changed daily so that in the ED tests, the fish and *Ceriodaphnia* were exposed to a new 24-hour composite effluent sample each day. The dilution water was a new daily grab sample of receiving water. For the ambient tests, only *Ceriodaphnia* were tested and they were placed in a new daily grab sample each day. The controls for the STP, refinery, and chemical plant ED tests were in the same water as the animals in the ambient tests for Stations 2, 3B, and 4, respectively.

For the fathead minnow larval growth tests, a chamber 30 cm x 15 cm x 10 cm deep was made and divided by three glass partitions which resulted in four compartments 13 cm x 7.6 cm x 10 cm deep. The partitions stopped 2.5 cm short of one side of the chamber and a piece of stainless steel screen was glued from one chamber end to the other and across the ends of each compartment. This left a narrow sump 2.5 cm x 30 cm x 10 cm deep along one side of the chamber to which each of the four compartments was connected by its screen end (Figure A-1). In this way, the compartments could be filled and drained by adding to or removing water from the sump, while the fish in the compartments remained relatively undisturbed. This design allowed four "replicates" for each concentration. These were not replicates in the pure statistical sense because there was a water connection between compartments. However, there was virtually no water movement between compartments except when the compartments were filled or drained.

Each day, 0.1 ml of newly hatched brine shrimp was fed three times and survival was counted. Live brine shrimp were available during the entire daylight period of 16 hours. Light intensity was very dim. The compartments were siphoned daily using a rubber "foot" on a glass tube to remove uneaten brine shrimp. Additional test solution was removed from the sump until about 500 ml remained in the four compartments combined, leaving a water depth of about 1 cm. Then, approximately 2,000 ml of new test solution was added slowly into the sump. The larval fish were able to maintain their position against the current easily while the chambers were filled. Fish were assigned to compartments one or two at a time in sequential order. They were less than 24 hours hatched at the test beginning and were obtained from the Newtown Fish Toxicology Laboratory culture unit. At the end of the test, the fish were rinsed in distilled water, oven-dried at 98°C for 18 hours, and weighed on an analytical balance. Four lots of 10 fish were preserved at the beginning of the test and later weighed to estimate initial weight. The method is further described in Norberg and Mount (in press).

The *Ceriodaphnia* from the Environmental Research Laboratory—Duluth (ERL-D) culture unit, were placed one animal to each of ten 30-ml beakers for each concentration or sample tested. Each beaker contained 15 ml of test water; a newly born *Ceriodaphnia*, less than 6 hours old, was added. One drop of a yeast

Figure A-1. Test chamber for static renewal fathead minnow larvae growth test.



food, containing 250 μg, was fed daily. Each day, the animal was moved to a new 15-ml volume with an eye dropper, and the yeast food again was added. When young were present, they were counted and discarded. Temperatures were maintained at 23-25°C. For the ED tests, the same concentration and change schedules were used as described for the fathead minnows. Mount and Norberg (1984) describe the culture procedures and test method with *Ceriodaphnia*.

Light was kept very dim to avoid algal growth and to keep conditions comparable to those used for culturing at Duluth. The high bacterial content of the water and waste samples increased available food and, where toxicity was not present, better young production was obtained than where the only food was the yeast, as was the case for the refinery test using Lake Superior water for dilution.

The median lethal concentrations (LC50s) were determined using the trimmed Spearman-Karber procedure (Hamilton et al. 1977). Significant differences among the LC50s were determined by one-way analysis of variance and Duncan's multiple range test.

A.2 1983 Methods

In 1983, one 19-liter grab sample was collected for each ambient station; 24-hour composite samples of each effluent were collected. The samples were cooled to 8°C and transported back to ERL-D where effluent dilution tests and ambient tests were run with both *Ceriodaphnia* and the fathead minnow. The fathead minnows were less than 24 hours hatched and were obtained from the ERL-D culture unit. Test and analyses procedures followed the description in Part 1. Temperature was maintained at 25±1°C during these tests.

Appendix B. Hydrological Field and Analytical Procedures

Dye was injected continuously for approximately 24 hours at each site to establish an equilibrium between the injection-point dye concentration and the downstream dye distribution. On the second day of each study, water samples were collected at 12 transects extending from 30 m above to 1,520 m below the point of discharge. The transect locations with respect to the three discharges are illustrated in Table 6-1. The ratio of the dye concentration at the point of discharge to the dye concentration in the water samples collected at the downstream transects represents the dilution undergone by the effluent.

Rhodamine WT dye was injected at each site by a Fluid Metering, Inc. precision metering pump. The injection system was placed at a sufficient distance from the river to allow complete mixing of the dye and effluent prior to the point of discharge. The weight of the dye container was periodically recorded to monitor the dye injection rate. The Rhodamine WT dye used in the study decays in the presence of chlorine. Sodium thiosulfate, $Na_2S_2O_3$, reduces the chlorine to chloride when present in a concentration approximately six times as great as the chlorine level. At the Lima STP a second Fluid Metering, Inc. precision metering pump injected a 400 gm/liter solution of $Na_2S_2O_3$.

A flow-through Turner Designs fluorometer was set up where the discharge from each site enters the river to provide a continuous record of discharge dye concentration. The fluorometer reading was recorded on a Rustrak strip chart recorder. The temperature at the discharge was recorded using a YSI probe and an Esterline Angus strip chart recorder because the fluorometer reading is temperature-dependent. Prior to the field survey, the two fluorometers used had been calibrated over a range of 0-124 ppb dye.

During the ambient survey on the second day of dye injection, water samples were collected in 200-ml bottles. A sample was taken and the water depth recorded every 3 m across the transect, except near a discharge or at a narrow transect where a 1.5-m interval was used for greater resolution. A manual sampler was set to take the water samples 0.2 m from the bottom. When the depth was less than 0.25 m, the sample was taken at middepth. If the water depth was greater than 0.5 m, a second sample was taken 0.1 m from the surface. Water samples were processed on the same day of the instream survey using a Turner Designs fluorometer in the discrete sample mode. The fluorometer calibration was checked with field standards each day it was used. As part of each ambient survey, a flow measurement was taken at a transect located 152 m upstream of the STP using a Teledyne Gurley Pygmy flowmeter. This upstream flow, coupled with the reported discharge flows, allowed the river flow to be calculated below each discharge.

The fluorometer data was converted to dye concentration, C(ppb), using the relationship

$$C(\text{ppb}) = SR\,\exp(0.027(T-20))$$

where

S = slope from the calibration regression for the appropriate sensitivity scale of the fluorometer

R = fluorometer reading

T = temperature of the grab sample at the time it was processed

exp(0.027(T-20)) = correction factor for the temperature dependence of fluorescence (20°C is the reference temperature)

In a similar fashion, the fluorometer readings from the discharge strip chart recorder were reduced every 30 minutes for the duration of the study. The background levels (equivalent dye concentration fluorescence) measured upstream of the discharge and in the effluent prior to dye injection were flow-weighted to determine a background level which was subtracted from the ambient data.

The 48-hour interval between collecting a set of water samples at each site was considered sufficient for the dye from the previous 24-hour injection to flush out of the current 1,500-m study area.

On 20 and 21 September, a dye integrity study was performed by adding Rhodamine WT dye to an effluent sample from each of the three sites. For each site a 50 ppb dye solution was made in order to represent the dye-injection concentration, and a 5 ppb dye solution was made by diluting a portion of the 50 ppb solution with upstream river water. The solution for the STP also contained sodium thiosulfate. Each solution was measured in the fluorometer immediately after mixing, one hour later, and one day later. No noticeable decay was observed in any of the samples.

Appendix C.
Biological Methods

C.1 Periphytic Community

Natural substrates (rocks) were sampled quantitatively using an epilithic algal bar-clamp sampler at each of nine stations. All samples were taken from the lower end of riffle areas and runs located at each station. Four replicate samples were taken at each station for chlorophyll *a* and biomass measurements. These samples were filtered using 0.45-μm filters and stored in ice to await analysis in the laboratory. One sample consisting of a composite of two bar-clamp collections was taken from each station for cursory identification (genus level) and abundance estimates. These samples were preserved in M3 preservative to await analysis.

Ash-free dry weights (AFDW) and chlorophyll *a* were analyzed from the filters in the laboratory. A small plug (of equal size) was removed from each filter for chlorophyll *a* analyses. The plugs of the filters were macerated, and chlorophyll *a* was extracted with a chlorophyll *a* standard (Sigma Chemicals) extracted in a 90 percent acetone solution. Chlorophyll *a* standing crop was expressed as milligrams per square meter (mg/m^2). For AFDW, the remaining portions of the filters were dried at 105°C to a constant weight and ashed at 500°C. Distilled water then was added to replace the water of hydration lost from clay and other minerals. Samples were redried at 105°C, and biomass standing crop was expressed in grams per square meter (g/m^2). The biomass and chlorophyll *a* data were used to calculate the Autotrophic Index (Weber 1973), which indicates the relative proportion of heterotrophic and autotrophic (photosynthetic) components in the periphyton. The biomass and chlorophyll *a* data were also statistically tested by analysis of variance (Steele and Torrie 1980) and multiple comparison tests to detect significant ($P \leq 0.05$) differences between sampling locations.

For identification and enumeration, each sample was mixed for 30 seconds in a blender to disrupt algal clumps, and the sample volume was then increased to 250 ml. Ten percent of each thoroughly mixed sample was removed to prepare Hyrax slides, which were examined at 1,250X magnification to confirm the identity of diatoms encountered during the quantitative analyses. A 0.1-ml aliquot from each quantitative sample was placed in a settling chamber designed for use on an inverted microscope. The chamber was then filled with deionized water, and periphytic forms were allowed to settle to the bottom of the chamber for 24 hours. Samples were examined at 1,000X magnification with an inverted microscope, and algae were identified to genus. For each example, one diameter of the counting chamber was examined, and algae containing protoplasm were enumerated as cells except for genera of filamentous blue-green algae, which were counted in 10-μm units of length. The actual number of cells identified and counted in each sample ranged from 126 to 2,536, but was greater than 400 in all but two samples. Periphyton abundance was expressed as number of cells per square millimeter ($cells/mm^2$), and genus diversity and equitability were calculated by USEPA methods (EPA 1973).

C.2 Benthic Macroinvertebrate Community

Benthic samples were collected from all nine stations in 1982 with a Hess stream sampler (881 cm^2). Five replicate samples were collected from the riffle habitat at each station. The mesh size on the Hess sampler is 363 μm, thereby retaining early instars of macroinvertebrate life stages. Samples were preserved in 10 percent buffered formalin and returned to the laboratory for analysis.

Water quality measurements consisting of temperature, dissolved oxygen, pH, and conductivity were taken at every station. The water quality for the biological field efforts are discussed in Section 4.1.

Drift collections were made at reference Station 2 and affected Stations 3, 4, and 5. The stream drift nets were 30.5 cm x 45.7 cm x 3.7 cm, made of 500-μm mesh nytex screen, and were anchored in the run areas of each station. Four replicate samples were collected from each station. Drift sampling was conducted after dark, and nets were left in place for 30 minutes. Velocity measurements were taken in front of each net with a Gurley Pygmy flow meter to enable quantification of the data.

During the 1983 survey, three quantitative samples were taken at the quarter points across the riffle at each station using a standard Hess sampler (881 cm^2

with a 800 x 900-μm mesh screen). A qualitative sample was taken by combining kick sampling from recognized different habitats using a dip net with 500-μm mesh. Benthic samples were transferred in total into glass jars and preserved in 10 percent formalin.

The benthic samples contained large amounts of detritus and organisms and were subsampled to expedite organism sorting and identification. Subsampling of the 1982 samples was done using EPA's pneumatic rotational sample splitter (patent pending). Samples were sorted with the aid of a Wild M-5 dissecting microscope. Organisms were sorted into major taxonomic categories and preserved in 70 percent alcohol for later identification; organisms were identified to the lowest practical taxon using appropriate keys and references. Oligochaetes and chironomid larvae were mounted on microslides prior to identification.

C.3 Fish Community

Fish collections during the 1982 survey were made in 92.3-m sections of stream at each of the nine Ottawa River stations. Each sampling area contained pool and riffle habitats, although in widely varying proportions (Table C-1). The riffle was upstream of the pool at all stations except 5 and 6. The riffles were considered natural barriers to the pool-dwelling fish and a block net was placed at the opposite end of each station to act as a barrier to escape.

Table C-1. Station Pool, Riffle Proportions, and Number of Seine Hauls

	Pool		Riffle	
Station	% of Station	Number Hauls	% of Station	Number Kick-seines
1	90	6	10	5
2	70	6	30	7
3	40	4	60	15
4	75	6	25	10
5[a]	40	5	60	18
6[a]	70	6	30	9
7	50	6	50	11
8	15	2	85	14
9	70	6	30	2

[a] Pool upstream of riffle.

The pools were sampled using either a 12-m or 13.8-m x 3.7-m deep bag seine with 0.3-cm mesh. A 10.2-m x 3.7-m deep straight seine with 0.3-cm mesh was used in the riffles employing the "kick seine" technique. The number of seine hauls or kick seines varied according to the width and other physical characteristics to ensure complete sampling of the area within the station.

Water temperature, dissolved oxygen, specific conductance, and pH were measured during fish collections at each station. A Hydrolab Model 4041 was used for all measurements.

During the 1983 survey, fish were collected using a 3.7-m x 27.7-m x 0.3-cm woven seine. Extensive sampling was conducted in all recognizable habitats. The small fish were killed in ice water and preserved in 10 percent formalin. The few large fish were identified in the field and released.

C.4 Fish Caging Study

The caging study was conducted using commercially available plastic minnow traps with the openings plugged with rubber stoppers. The maximum mesh size was 4 x 8 mm. Total volume of each cage was 10 liters. Three cages were used at each of six stations, and were labeled Rep A, B, and C, from downstream to upstream. Each cage was secured to the bank with a light line.

Fish used in the caging study were collected from two upstream locations (Thayer Road and Cool Road). The redfin shiner (*Notropis umbratilis*) was selected for its abundance and relative ease of identification with minimal handling stress. The fish were transported and held in a large stainless steel tank containing approximately 230 liters of water. At each station, 10 fish were transferred from the holding tank to the minnow trap contained in a 19-liter bucket filled with receiving water for transport to the caging site. To reduce stress at each handling, care was taken to move the fish quickly but gently in either a very fine mesh net or a small quantity of water cupped in the hand.

Observations were made daily at approximately the same time and the number of live fish recorded. Dead fish were removed and discarded. At the end of six days, all remaining live fish were removed and frozen for bioaccumulation analysis to be performed by EPA labs in Duluth.

C.5 Zooplankton

A Wisconsin stream plankton net with 80-μm mesh screen was used to collect zooplankton from each biological station in 1983. Water velocities were determined by timing the drift of a float (small leaf) over a 9.2-m measured distance. The net was exposed in this course for a duration of 2 minutes. Two replicate samples were collected consecutively at each station. Each sample was preserved with 10 percent formalin and stored in a 120-ml glass jar.

In the laboratory, the volume of each replicate sample was determined to the nearest ml with a 250-ml graduated cylinder. After mixing, 1 ml was transferred to Sedgewick-Rafter counting chamber and the total subsample strip scanned at 40X using a compound microscope. Two such subsamples were analyzed from each replicate sample.

Appendix D
Support Biological Data

Table D-1. **Abundance (cells/mm²) of Periphytic Algae on Natural Substrates in the Ottawa River, September 1982**

	Sampling Stations								
	1	2	3	4	5	6	7	8	9
BACILLARIOPHYTA (Diatoms)									
Centrales									
Cyclotella	3,392	9,279	499	1,596	100	1,895	200	399	499
Melosira	0	0	0	0	0	0	0	200	0
Skeletonema	0	1,895	299	698	100	0	0	0	399
Stephanodiscus	0	299	0	299	0	499	0	0	100
Thalassiosira	0	499	0	0	0	0	0	599	100
Total Centrales	3,392	11,972	798	2,593	200	2,394	200	1,198	1,098
Pennales									
Achnanthes	100	200	100	2,494	0	5,486	1,097	399	0
Amphora	499	2,893	200	299	0	0	0	499	0
Cocconeis	200	200	0	0	0	0	200	898	798
Gomphonema	0	0	0	1,097	0	499	200	0	0
Navicula	5,187	10,175	3,890	5,387	698	6,284	4,190	18,254	1,895
Nitzschia	3,192	7,382	499	28,928	1,197	9,875	24,239	9,377	3,491
Pinnularia	0	0	0	499	0	100	0	0	0
Rhoicosphenia	0	200	1,895	998	399	399	698	1,696	100
Surirella	100	0	0	0	0	100	0	0	0
Total Pennales	9,278	21,050	6,584	39,702	2,294	22,743	30,624	31,123	6,284
Total Bacillariophyta	12,670	33,022	7,382	42,295	2,494	25,137	30,824	32,321	7,382
CHLOROPHYTA (Green Algae)									
Nonfilamentous									
Ankistrodesmus	0	299	299	200	0	100	0	100	0
Chlamydomonas	0	0	100	200	200	299	100	0	100
Chlorella	0	0	0	0	0	0	599	0	0
Lagerheimia	0	100	0	0	0	0	0	0	0
Lobomonas	0	100	0	0	0	0	0	0	0
Oocystis	0	0	0	100	0	0	0	0	0
Pediastrum	0	0	0	0	0	0	0	0	2,993
Scenedesmus	598	798	200	2,394	200	100	798	399	499
Schroederia	0	0	0	299	0	100	100	100	0
Selenastrum	0	0	100	0	0	0	0	0	0
Total Nonfilamentous	598	1,297	699	3,193	400	599	1,597	599	3,592
Filamentous									
Cladophora	0	0	0	0	0	698	0	100	0
Stigeoclonium[a]	3,193	12,070	244,789	5,886	52,170	61,247	8,878	5,586	698
Total Filamentous	3,193	12,070	244,789	5,886	52,170	61,945	8,878	5,686	698
Total Chlorophyta	3,791	13,367	245,488	9,079	52,570	62,544	10,475	6,285	4,290
CRYPTOPHYTA (Cryptomonads)									
Chroomonas	0	100	0	0	0	0	0	0	0
Total Cryptophyta	0	100	0	0	0	0	0	0	0

Table D-1. (Continued)

	Sampling Stations								
	1	2	3	4	5	6	7	8	9
CYANOPHYTA (Blue-green Algae)									
Nonfilamentous									
Chroococcus	0	1,995	0	0	0	0	0	0	0
Merismopedia	0	798	0	0	0	0	0	0	0
Unidentified coccoids	0	2,195	0	0	0	0	0	0	0
Total Nonfilamentous	0	4,988	0	0	0	0	0	0	0
Filamentous[b]									
Lyngbya	1,995	2,294	0	1,496	0	599	2,893	1,796	898
Oscillatoria	100	1,397	0	2,793	0	1,596	2,095	0	0
Total Filamentous	2,095	3,691	0	4,289	0	2,195	4,988	1,796	898
Total Cyanophyta	2,095	8,679	0	4,289	0	2,195	4,988	1,796	898
EUGLENOPHYTA (Euglenoids)									
Euglena	0	0	100	100	0	0	0	0	0
Total Euglenophyta	0	0	100	100	0	0	0	0	0
Total Periphyton	18,556	55,168	252,970	55,763	55,064	89,876	46,287	40,402	12,570

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[a] *Stigeoclonium* included both basal and filamentous cells, as well as coccoid green algae that may possibly be growth forms or life stages of this genus.

[b] Filamentous blue-green algae were counted in 10-μm units of length.

Table D-2. Chlorophyll *a* and Biomass Data and Statistical Results for Periphyton Collected from Natural Substrates in the Ottawa River, September 1982

Parameter	Sampling Stations								
	1	2	3	4	5	6	7	8	9
Chlorophyll *a* (mg/m^2)									
Rep 1	13.3	326.9	226.8	260.0	160.0	58.8	170.0	—	59.4
Rep 2	40.6	230.5	316.6	150.0	140.0	240.0	230.0	228.0	180.0
Rep 3	41.8	210.0	130.0	115.0	100.0	180.0	175.0	57.6	50.3
Rep 4	—	326.9	512.5	250.0	46.7	125.0	91.5	122.0	120.0
Mean	31.9	273.6	296.5	193.8	111.7	151.0	166.6	135.9	102.4
Biomass (g/m^2)									
Rep 1	23.2	71.2	33.8	28.9	18.0	14.6	20.6	11.5	22.8
Rep 2	28.8	65.7	30.7	17.7	17.6	47.4	24.5	35.7	30.3
Rep 3	19.2	46.5	24.3	38.6	25.0	57.4	22.8	15.9	8.6
Rep 4	17.8	63.7	54.8	34.2	17.5	38.8	10.7	29.7	13.2
Mean	22.2	61.8	35.9	29.8	19.5	39.6	19.6	23.2	18.7
Autotrophic Index (Weber 1973)	971	230	135	176	216	269	119	225	208
Statistical Results: [a]									
Chlorophyll *a*									
$F = 6.990$ Station[b]	1	9	5	8	6	7	4	3	2
$P < 0.001$ Mean[c]	3.342	4.496	4.616	4.762	4.894	5.064	5.209	5.572	5.790
Biomass									
$F = 4.371$ Station	9	7	5	8	1	4	3	6	2
$P < 0.003$ Mean	2.818	2.930	2.960	3.044	3.085	3.356	3.535	3.562	4.111

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[a] Results based on analysis of variance and Tukey multiple comparison test performed on data transformed with natural logarithms [ln (x + 1)]. Stations underscored by a continuous line were not significantly different ($P > 0.05$).

[b] Stations are listed in order of increasing mean values.

[c] Means of transformed data.

Table D-3. Ranked Abundance Listing of All Macroinvertebrates Collected from Ottawa River, 21 September 1982

Species Name	Number	Percent	Cumulative Percent
Simuliidae, L.	5359.617	24.587	24.587
C. (Cricotopus) Bicinctus GRP.	4857.687	22.285	46.872
Cheumatopsyche, L.	2435.019	11.171	58.042
Hydropsyche, L.	1161.009	5.326	63.368
Thienemannimyia, GRP.	1041.965	4.780	68.148
Hydropsychidae, L.	913.723	4.192	72.340
Stenelmis L.	769.451	3.530	75.870
Simuliidae, P.	514.019	2.358	78.228
C. (Cricotopus) Tremulus GRP.	455.942	2.092	80.319
Chironomidae, P.	423.619	1.943	82.263
Elmidae, L.	383.149	1.758	84.021
Baetis, N.	318.765	1.462	85.483
Empididae, L.	309.830	1.421	86.904
Polypedilum (S.S.) Convictum, L.	221.007	1.014	87.918
Nanocladius, L.	203.400	0.933	88.851
Caenis, N.	188.421	0.864	89.715
Stenelmis, A.	167.923	0.770	90.486
Polypedilum (S.S.) Scalaenum, L.	153.207	0.703	91.189
Bothrioneurum Vejdovskyanum	148.214	0.680	91.869
Ephemeroptera, N.	146.112	0.670	92.539
IMM Tubif with Cap Chaet	142.695	0.655	93.193
IMM Tubif w/o Cap Chaet	121.409	0.557	93.750
Diptera, P.	107.744	0.494	94.245
Berosus, L.	98.809	0.453	94.698
Tricladida	98.284	0.451	95.149
Chironomus, L.	81.728	0.375	95.524
Physella	73.056	0.335	95.859
Baetidae, N.	62.807	0.288	96.147
Cricotopus Syvlestris GRP., L.	60.705	0.278	96.426
Heptageniidae, N.	49.142	0.225	96.651
Rheotanytarsus, L.	45.200	0.207	96.858
Limnodrilus Udekemianus	42.047	0.193	97.051
Stenacron, N.	39.419	0.181	97.232
Glyptotendipes, L.	38.367	0.176	97.408
Tanytarsus, L.	37.579	0.172	97.580
Hydroptila, L.	36.528	0.168	97.748
Simulium, L.	34.951	0.160	97.908
Potamothrix Bavaricus	32.586	0.149	98.058
Ancylidae	31.798	0.146	98.204
Piguetiella Michiganensis	26.805	0.123	98.327
Microtendipes, L.	22.600	0.104	98.430
Heptageniinae, N.	20.498	0.094	98.524
Nais Variabilis	18.921	0.087	98.611
Dero (Dero) Digitata	18.658	0.086	98.697
Psephenus, L.	17.870	0.082	98.779
Hydroptilidae, L.	16.819	0.077	98.856
Dicrotendipes L.	16.556	0.076	98.932
Sphaerium	14.979	0.069	99.001
Trichoptera, L.	11.563	0.053	99.054
Thienemanniella, L.	11.563	0.053	99.107
Stenonema, N.	11.300	0.052	99.159
Labrundinia, L.	11.300	0.052	99.210
Ablabesmyia, L.	10.512	0.048	99.259
Dero Funcata	9.986	0.046	99.304
Caenidae N.	9.460	0.043	99.348
Tubifex Tubifex	8.147	0.037	99.385
Cricotopus Tibialis, L.	7.358	0.034	99.419
Turbellaria	6.833	0.031	99.450
Gastropoda	6.307	0.029	99.479
Nais Communis	6.307	0.029	99.508
Branchiobdellidae	6.307	0.029	99.537
Potamanthus, N.	6.307	0.029	99.566
Argia, N.	5.256	0.024	99.590
Astacidae	4.467	0.020	99.611
Aulodrilus Pigueti	4.205	0.019	99.630
Tricorythidae, N.	4.205	0.019	99.649
Hemiptera U.	4.205	0.019	99.668
Cricotopus, P.	4.205	0.019	99.688

Table D-3. (Continued)

Species Name	Number	Percent	Cumulative Percent
Thienemaniella Nr. Fusca, L.	4.205	0.019	99.707
Corynoneura, L.	3.679	0.017	99.724
P. (Phaenopsectra) L.	3.153	0.014	99.738
Pyralidae, L.	2.628	0.012	99.750
Parachironomus Abortivus Type, L.	2.628	0.012	99.762
Limnodrilus Hoffmeisteri	2.365	0.011	99.773
Nais Bretscheri	2.365	0.011	99.784
Hyalella Azteca	2.365	0.011	99.795
Coenagrionidae, N.	2.365	0.011	99.806
Rhagovelia, A.	2.365	0.011	99.817
Chimarra, L.	2.365	0.011	99.828
Procladius L.	2.365	0.011	99.838
Cryptochironomus L.	2.365	0.011	99.849
Paratendipes Albimanus Type, L.	2.365	0.011	99.860
Chaetogaster Cristallinus	2.102	0.010	99.870
Pristina Longiseta Leidy	2.102	0.010	99.879
Crangonyx	2.102	0.010	99.889
Zygoptera, N.	2.102	0.010	99.899
Tortricidae, L.	2.102	0.010	99.908
Glossosomatidae, L.	2.102	0.010	99.918
Cricotopus Trifascia, L.	2.102	0.010	99.928
Eukiefferiella, L.	2.102	0.010	99.937
Paratanytarsus, L.	2.102	0.010	99.947
Collembola U.	1.577	0.007	99.954
Orconectes	1.314	0.006	99.960
Helisoma	1.051	0.005	99.965
Enchytraeidae	1.051	0.005	99.970
Elmidae, A.	1.051	0.005	99.975
Berosus, A.	1.051	0.005	99.979
Pseudochironomus L.	1.051	0.005	99.984
Nais Pardalis	0.526	0.002	99.987
Wapsa Mobilis	0.526	0.002	99.989
Rheocricotopus, L.	0.526	0.002	99.992
Pelecypoda	0.263	0.001	99.993
Naididae	0.263	0.001	99.994
Haemopis	0.263	0.001	99.995
Paraleptophlebia, N.	0.263	0.001	99.996
Hemiptera N.	0.263	0.001	99.998
Elodes, L.	0.263	0.001	99.999
Culicidae, P.	0.263	0.001	100.000

Note: L. = Larva
P. = Pupa
N. = Nymph
A. = Adult
U. = Unidentified
S.S. = *Sensu strictu* (in the strict sense)
Capitalization of taxa is due to computerized format.

Table D-4. **Shannon-Wiener Diversity Indices ($\bar{d}$), Associated Evenness and Redundancy Values, and Community Loss Indices Calculated on Benthic Data from Ottawa River, 1982**

Station	Diversity[a]	Evenness	Redundancy	Maximum Diversity	Minimum Diversity	Number of Species	Number of Individuals	Community Loss Index [b]
1	3.8950	0.6800	0.3212	5.7279	0.0242	53	7,124	—
2	3.6946	0.6481	0.3525	5.7005	0.0106	52	17,094	0.4651
3	1.7533	0.3507	0.6497	5.0000	0.0030	32	38,808	0.9643
4	2.3552	0.5072	0.4940	4.6439	0.0106	25	7,537	1.3636
5	1.7441	0.3554	0.6451	4.9060	0.0042	30	25,308	1.0769
6	2.3329	0.4625	0.5384	5.0444	0.0082	33	16,971	0.9667
7	2.2555	0.4210	0.5795	5.3576	0.0047	41	39,595	0.7500
8	3.6047	0.6495	0.3511	5.5546	0.0100	47	16,383	0.5000
9	2.4304	0.5111	0.4892	4.7549	0.0032	27	29,946	1.0435

[a] Calculated on a log base 2.

[b] Calculated using Station 1 as reference station.

Table D-5. **List of Fish Species and Families Collected from the Ottawa River Near Lima, Ohio, 24-26 September 1982[a]**

Family	Scientific Name	Common Name
Clupeidae (herring)	*Dorosoma cepedianum*	Gizzard shad
Cyprinidae (minnow)	*Cyprinus carpio*	Carp
	Notemigonus crysoleucas	Golden shiner
	Pimephales promelas	Fathead minnow
	Semotilus atromaculatus	Creek chub
	Notropis spilopterus	Spotfin shiner
	N. atherinoides	Emerald shiner
	Pimephales notatus	Bluntnose minnow
	Campostoma anomalum	Stoneroller
	Notropis stramineus	Sand shiner
	N. umbratilis	Redfin shiner
Catastomidae (sucker)	*Catastomus commersoni*	White sucker
	Moxostoma duquesnei	Black redhorse
	M. erythrurum	Golden redhorse
Ictaluridae (catfish)	*Ictalurus catus*	White catfish
	Noturus gyrinus	Tadpole madtom
Cyprinodontidae (killifish)	*Fundulus notatus*	Blackstripe topminnow
Centrarchidae (sunfish)	*Ambloplites rupestris*	Rock bass
	Lepomis cyanellus	Green sunfish
	L. macrochirus	Bluegill
	Micropterus salmoides	Largemouth bass
	Pomoxis annularis	Black crappie
	Lepomis x Lepomis	Sunfish hybrid
Percidae (perch)	*Etheostoma blennioides*	Greenside darter
	E. caeruleum	Rainbow darter
	E. flabellare	Fantail darter
	E. nigrum	Johnny darter
	Percina maculata	Blackside darter

[a] Names follow Robins et al. 1980.

Table D-6. Shannon-Wiener Diversity Indices ($\bar{d}$), Associated Evenness and Redundancy Values, and Community Loss Indices Calculated on Fisheries Data from Ottawa River, 1982

Station	Diversity[a]	Evenness	Redundancy	Maximum Diversity	Minimum Diversity	Number of Species	Number of Individuals[b]	Community Loss Index[c]
1	1.6693	0.4511	0.5503	3.7005	0.0093	13	20,295	—
2	0.9886	0.2530	0.7480	3.9069	0.0055	15	42,884	0.2667
3	0.4478	0.1294	0.8716	3.4594	0.0041	11	41,031	0.5455
4	1.8444	0.7135	0.3058	2.5850	0.1633	6	295	1.5000
5	1.3765	0.8685	0.1448	1.5850	0.1450	3	114	3.6667
6	—	—	—	—	—	0	0	12.0000
7	2.7956	0.9319	0.0757	3.0000	0.2996	8	214	1.1429
8	1.1320	0.3272	0.6744	3.4594	0.0086	11	18,194	0.8000
9	2.3278	0.5480	0.4552	4.2479	0.0294	19	8,917	0.3125

[a] Calculated on a log base 2.
[b] Abundance adjusted to number per 465 m^2 (sampling area).
[c] Calculated using Station 1 as reference station.

*USGPO: 1984-759-102-10650